







EFECTOS SOBRE LA AVIFAUNA DE LAS PLANTACIONES FORESTALES JÓVENES EN CAMPOS AGRÍCOLAS MEDITERRÁNEOS

Juan Salvador Sánchez Oliver

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Tesis Doctoral

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HACE CONSTAR:

Que el trabajo descrito en la presente memoria, titulado "**Efectos sobre la avifauna de las plantaciones forestales jóvenes en campos agrícolas mediterráneos**", ha sido realizado bajo su dirección por D. Juan Salvador Sánchez Oliver dentro del Programa de Doctorado 'Ecología. Conservación y Restauración de Ecosistemas' (D330), adscrito al Departamento de Ciencias de la Vida – Unidad Docente de Ecología de la Universidad de Alcalá. Esta tesis reúne los requisitos propios de este tipo de trabajo: rigor científico, aportaciones novedosas y aplicación de una metodología adecuada. Por lo tanto, doy mi Visto Bueno a la presentación de dicha Tesis Doctoral.

Alcalá de Henares, a 24 de Septiembre de 2013

Fdo.: Dr. José María Rey Benayas

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Alcalá de Henares, a 27 de Septiembre de 2013

Fdo.: Dr. Gonzalo Pérez Suárez

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Memoria presentada para optar al grado de Doctor por la Universidad de Alcalá

Programa de Doctorado:
'Ecología. Conservación y Restauración de Ecosistemas'
(D330)

Director: Dr. José María Rey Benayas

Co-director: Dr. Luis María Carrascal De La Puente

Alcalá de Henares, Septiembre de 2013

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Y hasta el día de hoy.

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A disfrutar de esta tesis y de lo que venga.

Foto: Búho chico (*Asio Otus*)

Autor: Juan S. Sánchez Oliver

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Foto: Alcaudón real (*Lanius meridionalis*)

Autor: Juan S. Sánchez Oliver

Resumen general

Resumen

Esta Tesis investiga los efectos de las plantaciones forestales jóvenes (<20 años) en campos agrícolas de ambientes mediterráneos promovidas por la PAC sobre las comunidades de aves que colonizan estas plantaciones y los hábitats agrícolas circundantes. Los objetivos específicos son: (1) caracterizar la composición y la diversidad de la comunidad de aves que coloniza el interior de las plantaciones forestales y establecer sus relaciones con las características del hábitat local y del paisaje circundante; (2) comprobar la influencia del muestreo aleatorio, las preferencias de hábitat y algunos rasgos autoecológicos en la aparición de las especies de aves en estas plantaciones; (3) evaluar el efecto de las plantaciones forestales jóvenes sobre la comunidad de aves propia de los medios agrícolas mediterráneos; y (4) examinar el papel de las plantaciones forestales en la depredación de nidos de aves tanto en las propias plantaciones como en los hábitats agrícolas adyacentes. Un fin aplicado de estas investigaciones es proporcionar propuestas para mejorar la gestión de las plantaciones forestales.

Para cumplir estos objetivos se realizaron tres estudios observacionales (objetivos 1-3) y otro experimental de depredación de nidos artificales (objetivo 4) en el Campo de Montiel (Ciudad Real) entre los años 2011 y 2013. Se caracterizó tanto el hábitat local (área, perímetro, conectividad y estructura de la vegetación de las plantaciones) como el paisaje circundante (proporción de los diferentes tipos de uso del suelo) y se censaron las aves en 61 plantaciones y en 40 campos de cultivo adyacentes a las mismas. También se comparó la composición y la abundancia de estas comunidades con la avifauna regional en torno al área de estudio.

Los resultados principales indican que (a) las comunidades de aves que ocupan el interior de las plantaciones forestales y los medios agrícolas cercanos a éstas y la depredación de nidos de aves son afectadas tanto por las características de la plantación como por las del paisaje circundante; (b) la hipótesis del muestreo aleatorio explica, principalmente, la presencia de especies de aves en las plantaciones, de manera que son colonizadas por especies de elevada amplitud de hábitat, con tendencias poblacionales positivas y de amplia distribución regional, pero no favorecen a especies forestales especialistas; (c) la comunidad de aves de zonas agrícolas alrededor de las plantaciones está influida negativamente por ellas, aunque con efectos distintos en el invierno y durante el periodo reproductor, y aparentemente atenuados por la heterogeneidad del paisaje; (d) las plantaciones forestales resultan en elevadas tasas de depredación tanto en su interior como en los campos de cultivo adyacentes, especialmente en paisajes homogéneos dominados por cultivos herbáceos; y (e) estas plantaciones forestales no promueven un incremento de la diversidad regional de aves, por lo que las ayudas de la PAC a la reforestación de tierras agrícolas no constituyen una práctica beneficiosa para la biodiversidad de este grupo taxonómico.

Los resultados obtenidos señalan el escaso valor de las plantaciones forestales basadas en pinos para la restauración de paisajes agrícolas mediterráneos y apuntan las siguientes recomendaciones para su gestión: (a) considerar el origen biogeográfico de la avifauna para la restauración forestal; (b) excluir, e incluso extirpar, la reforestación de tierras agrícolas en zonas con elevado interés para la conservación de las aves características de medios abiertos; (c) manejar las plantaciones existentes con actuaciones de clareo y poda; (d) controlar las poblaciones de depredadores generalistas mediante una gestión integral del mosaico agrícola y forestal; y (e) restaurar parches de vegetación leñosa semi-natural.

Palabras clave: Aves de espacios agrícolas abiertos – Composición de especies – Conservación – Efecto de la distancia – Especies forestales generalistas – Hábitat local – Índice SPEC – Muestreo aleatorio – Nidos artificiales – Preferencias de hábitat – Riqueza local de especies – Paisaje – Tasas de depredación – Usos del suelo – Variación estacional

Abstract

This PhD thesis investigates the effects of young tree plantations (<20 years) on Mediterranean former cropland motivated by the Common Agrarian Policy on bird communities inhabiting these plantations and the surrounding farmland habitat. The specific objectives are: (1) to characterize bird community composition and diversity in the interior of tree plantations and to establish their relationships with local habitat and surrounding landscape characteristics; (2) to test the influence of random sampling effects, habitat preferences and life history traits on bird species occurrence; (3) to assess the effect of young tree plantations on bird communities inhabiting Mediterranean farmland; and (4) to examine the role of tree plantations on bird nest predation at both tree plantations and adjacent farmland habitat. An applied aim of this research is providing recommendations to improve tree plantation management.

To achieve these objectives we carry out three observational studies (objectives 1-3) and an experimental study of artificial nest predation (objective 4) in Campo de Montiel (Ciudad Real) between 2011 and 2013. We characterized both local habitat (area, perimeter, connectivity, and vegetation structure of tree plantations) and surrounding landscape (proportion of different land-use types), and surveyed birds at 61 tree plantations and 40 adjacent cropland fields. We also compared composition and abundance of these communities with the regional avifauna around the study area.

Main results indicate that (a) bird communities inhabiting the interior of tree plantation and adjacent cropland and bird nest predation are affected by the characteristics of both tree plantations and surrounding landscape; (b) the random sampling hypothesis explains principally the occurrence of bird species in tree plantations, so they are colonized by species with broad habitat preferences, increasing population trends and widespread distribution, but do not favour specialist forest species; (c) bird community at cropland habitat adjacent to tree plantations is influenced negatively, although with different effects in the winter and the breeding season, which are apparently mitigated by landscape heterogeneity; (d) tree plantations result in high predation rates at their interior and on adjacent farmland habitat, specially in homogeneous landscapes dominated by herbaceous cropland; and (e) these tree plantations do not increase bird regional diversity, so CAP aids to cropland afforestation is not a beneficial practice for biodiversity of this taxonomic group.

The results indicate the low value of tree plantation based on pines for restoration of Mediterranean farmland and point the following recommendations for their management: (a) to consider the biogeographic origin of avifauna for forest restoration; (b) to exclude, and even to extirpate, tree plantations in agricultural landscapes that are highly valuable for open farmland bird species; (c) to thin and prune tree plantations; (d) to control the populations of generalist predators by means of integral management of the agricultural and forest mosaic; and (e) to restore patches of semi-natural woody vegetation.

Keywords: Artificial nests – Conservation – Distance effect – Habitat preference – Generalist forest species – Land use – Landscape – Local species richness – Local habitat – Open farmland birds – Predation rates – SPEC index – Random sampling – Seasonal variation – Species composition

Foto: Avutardas comunes (*Otis tarda*) alzando el vuelo sobre un viñedo, pastos y cultivo herbáceo; se observa una plantación forestal joven al fondo.

Autor: Juan S. Sánchez Oliver

Capítulo 1

Introducción general

Agricultura y biodiversidad

Más del 75% de las tierras emergidas y sin hielo permanente muestra alteraciones como consecuencia de la acción humana debidas a diferentes usos del suelo, mientras que menos de un 25% son medios naturales no transformados, principalmente zonas desérticas (Ellis & Ramankutty, 2008). La agricultura es la actividad humana con mayor ocupación del territorio, extendiéndose sobre aproximadamente el 37,8% de la superficie terrestre (FAO, 2013a).

Pocas actividades humanas tienen un carácter tan paradójico en lo referente a la conservación de la biodiversidad como la agricultura (Rey Benayas, Bullock & Newton, 2008). Por ejemplo, la agricultura es la principal causa de amenaza de aves en el mundo (BirdLife International, 2004a). Sin embargo, los agrosistemas pueden tener un importante valor para la conservación, particularmente en algunas regiones de Europa. Así, el 31% de los sitios Natura 2000 son manejados agrícolamente y algunas especies de especial interés en conservación dependen directamente de los sistemas agrícolas extensivos (BirdLife International, 2004a; Madroño, González & Atienza, 2004; Rey Benayas & Bullock, 2012).

En Europa se está produciendo actualmente la coexistencia de dos cambios en los usos del suelo que están afectando de manera relevante a la biodiversidad: el abandono de tierras agrícolas, algunas de las cuales han sido o están siendo reforestadas (ver más adelante), y la intensificación de los cultivos. Así, desde el año 1960, la superficie agrícola total ha pasado de 212.301.000 ha a 186.555.550 ha, es decir, ha disminuido un 12,1% (FAO, 2013a). Durante este periodo, la superficie agrícola de regadío, que puede considerarse una variable indicadora de la intensificación del uso agrícola, ha aumentado de 9.207.000 ha a 17.937.040 ha, es decir, se ha incrementado del 4,3% al 9,6% del total de la superficie agrícola (FAO, 2013a) (**Figura 1**).

Las aves constituyen un grupo relativamente bien estudiado con reconocido valor indicador que puede aportar información para mejorar la gestión de diversos hábitats Los agrosistemas son importantes para el mantenimiento de la diversidad de aves de Europa, especialmente para las aves con importancia en conservación (Delgado & Moreira, 2000; BirdLife International, 2004b). La intensificación y el abandono de la agricultura son señaladas como amenazas directas para las especies que dependen de los medios agrícolas extensivos tradicionales y que, de acuerdo a la legislación europea actual, constituyen un grupo con especial interés

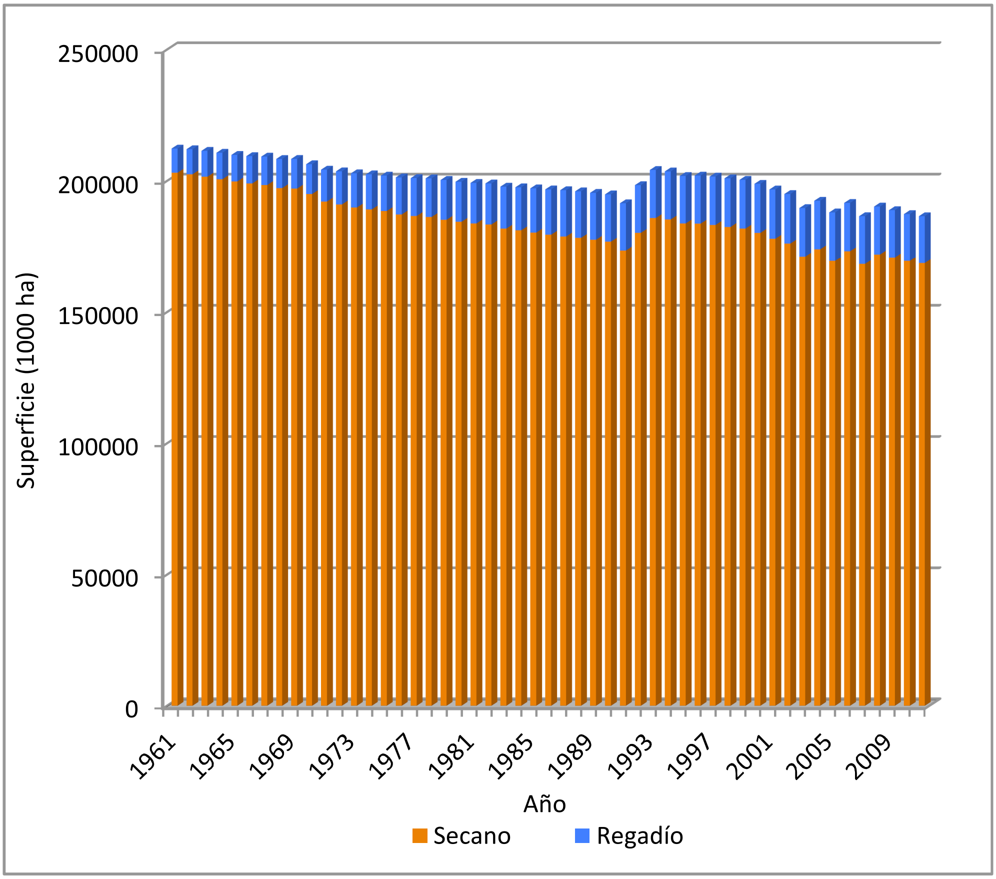

Figura 1. Cambios en la superficie de cultivos de secano y cultivos de regadío en Europa +27 desde el año 1960 hasta el año 2011 (FAO, 2013a).

en conservación (Sirami, Brotons & Martin, 2007; Vallecillo, Brotons & Herrando, 2007). Según datos de la European Bird Census Council (2010), en el periodo comprendido entre 1980 y 2007 ha habido una tendencia negativa de las aves comunes características de los medios agrícolas, produciéndose una merma promedio del 35% en las poblaciones de las mismas en cuatro regiones de Europa (**Figura 2**). El Índice de Aves Agrícolas Comunes, "un barómetro de cambio en la biodiversidad de tierras agrícolas en Europa", muestra un declive en estas poblaciones de aves de casi el 20% entre 1990 y 2008 (Directorate-General for Agriculture and Rural Development, 2012; ver también, Gregory *et al.*, 2005; Butler *et al.*, 2010; Guerrero *et al.*, 2012).

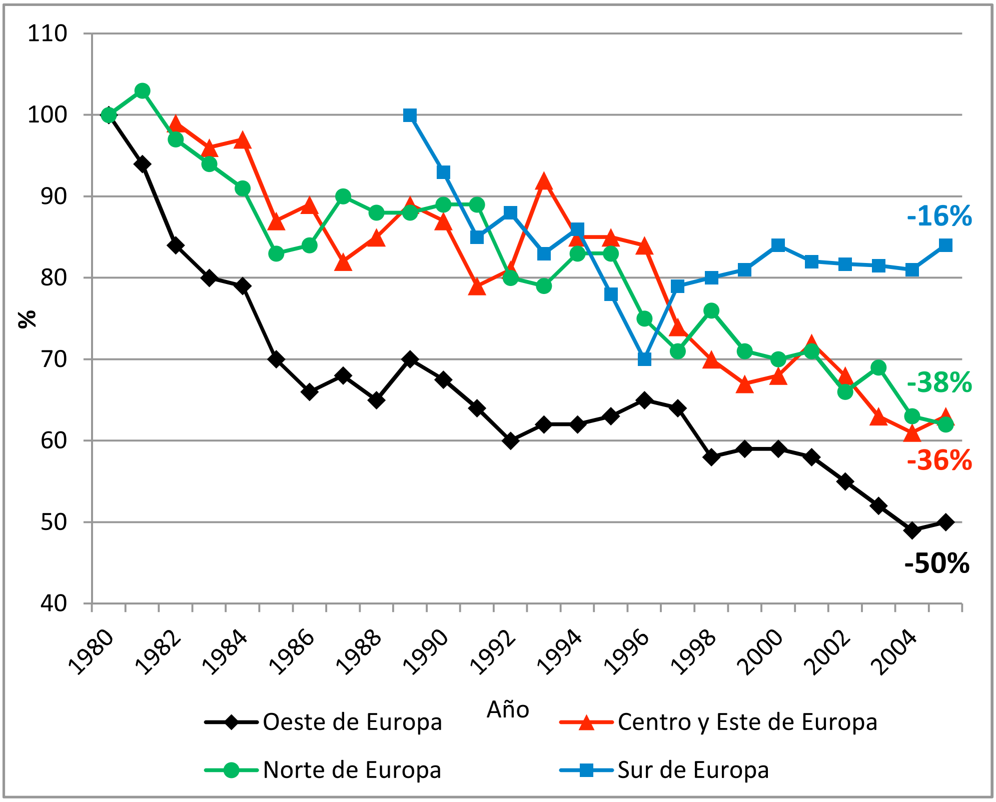

Figura 2. Tendencias de cambio de las poblaciones de aves agrícolas comunes en cuatro regiones diferentes de la Unión Europea (Oeste, Este, Norte y Sur) desde 1980 hasta 2005. A la derecha de cada tendencia se muestra el porcentaje de declive (European Bird Census Council, 2013).

Reforestación pasiva y plantaciones forestales

Las áreas agrícolas que dejan de ser cultivadas son objeto de la sucesión secundaria o restauración pasiva de la vegetación, o bien pueden acogerse a programas de reforestación activa o plantaciones forestales. Ambos procesos, restauración pasiva y restauración activa, provocan cambios netos en la superficie forestal de cada país. A nivel global, en la década comprendida entre los años 2000 y 2010, la reforestación pasiva ha recuperado 2,9 millones ha/año de bosque secundario y las plantaciones forestales se han incrementado en 4,9 millones ha/año (FAO, 2011; Rey Benayas & Bullock, 2012). Actualmente, un 7% de la superficie forestal en el planeta son bosques plantados. A pesar de ello, la superficie forestal mundial se ha reducido en un 3,4% en las dos últimas décadas

(FAO, 2011). Sin embargo, en Europa, la superficie forestal ha aumentado debido, sobre todo, al abandono de la actividad agrícola (**Figura 1**) y la migración de la población rural a las ciudades. Desde el año 1990, la superficie forestal ha aumentado un 19,7%, constituyendo ahora el 37,6% de la superficie terrestre europea (FAO, 2013a). Es previsible que esta superficie forestal aumente en un futuro próximo debido al fenómeno socioeconómico de la migración rural anteriormente explicado (**Figura 3**; Navarro & Pereira, 2012) y que una fracción de las tierras agrícolas abandonadas sea sujeto de reforestación activa.

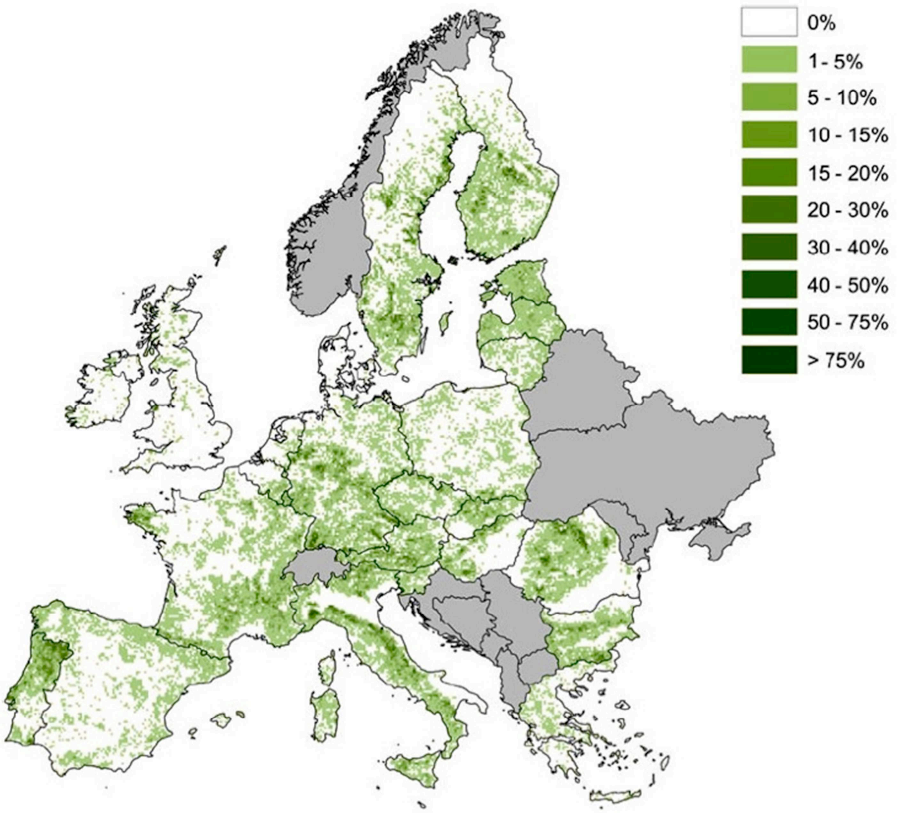

Figura 3. Proyección de la distribución de los puntos calientes para la biodiversidad ("*hotspots*") en tierras agrícolas renaturalizadas, es decir, abandonadas y reforestadas de forma pasiva, en el año 2030. Estos *hotspots* son áreas clasificadas como agrícolas en el año 2000. Los *hotspots* aparecen expresados como el porcentaje que ocupan en cada cuadrícula de 10 km^2. Las áreas agrícolas del punto de partida corresponden a las categorías tierras arables de secano, pastos, cultivos de regadío y cultivos permanentes según la clasificación de FAO (2013b). Las áreas renaturalizadas corresponden a las categorías de vegetación semi-natural, bosques, tierras arables abandonadas recientemente y pastos abandonados recientemente según la anterior clasificación. Los países en gris no tienen datos (extraído de Navarro & Pereira, 2012).

Además, en Europa se han ejecutado reforestaciones de tierras agrícolas en el marco de la Política Agraria Comunitaria (PAC) definida por la Directiva de la CEE Nº 2080/92 de 1992. Estos programas de reforestación persiguen tanto beneficios sociales como ambientales, incluyendo el control de la erosión, la prevención de la desertificación, la regulación de los regímenes hídricos, el incremento de las tasas de fijación de dióxido de carbono y la restauración de antiguos hábitats forestales y su biodiversidad asociada (Comunidad Económica Europea, 1992). Así, el primer programa de ayudas a la reforestación en tierras agrícolas, incentivó la reforestación de 519.350 ha entre 1993 y 1997, de las que 238.901 ha (46%) fueron en España; el segundo programa, de 2000 a 2006, reforestó más de 6.500.000 ha en Europa incluyendo más de 200.000 ha en España; y el tercer programa, que comenzó en 2007 y sigue vigente, ha reforestado casi 1.200.000 ha en Europa y unas 100.000 ha en España. En total, las ayudas de la PAC han favorecido la reforestación de más de 8.000.000 ha de tierras agrícolas, la mayor parte de ellas en el sur de Europa (European Commision, 2013a, 2013b). En el caso de España, con 18.349.300 ha de áreas boscosas que constituyen el 36,8% de la superficie terrestre, la superficie forestal se incrementó en 176.000 ha/año durante el periodo 2005-2010 (FAO, 2010). Un 85% del área total de los bosques españoles corresponde a bosques regenerados naturalmente, y el restante 15% corresponde a bosques plantados (FAO, 2010).

El debate en torno a las plantaciones

Las plantaciones forestales son un ejemplo de ecosistemas noveles e híbridos según Hobbs *et al.* (2009), es decir, ecosistemas creados por el hombre que son diferentes a los ecosistemas históricos. Por ejemplo, las reforestaciones en campos de cultivo en el sur de Europa están basadas principalmente en especies de coníferas como *Pinus halepensis*, *P. pinaster* y *P. pinea*, aunque se emplean otras especies como *Retama sphaerocarpa* y *Quercus rotundifolia*. Estas especies son nativas en España peninsular, pero las plantaciones de coníferas en zonas agrícolas constituyen hábitats forestales muy diferentes a los que la restauración pasiva hubiera producido.

Las plantaciones forestales en tierras agrícolas producen una serie de beneficios sociales, económicos y ambientales pero, sin embargo, pueden tener efectos notables sobre las comunidades biológicas. Así, Bremer & Farley (2010) encontraron que las plantaciones forestales contribuían más a la biodiversidad cuando eran establecidas en áreas degradadas en lugar de reemplazar ecosistemas naturales y cuando se usan especies de árboles autóctonas en lugar de especies alóctonas. De modo similar, un meta-análisis de la riqueza y la composición de especies de flora y fauna en plantaciones madereras y pastizales en 36 sitios diferentes del mundo concluye que las plantaciones soportan mayores riquezas o abundancias de especies que los pastizales, solamente para grupos taxonómicos concretos (por ejemplo, herpetofauna) o con características del paisaje específicas (por ejemplo, ausencia de vegetación remanente en los pastizales) Las plantaciones forestales en paisajes agrícolas homogéneos aumentan la fragmentación de los hábitats cultivados (Fischer & Lindenmayer, 2007). Esta fragmentación puede tener diferentes efectos dependiendo de los tipos de cultivos que dominan el paisaje (Fahrig, 2003). Las plantaciones forestales en campos de cultivo causan pérdida de hábitat agrícola en cantidad y en calidad. Esta disminución de hábitat agrícola y el cambio en el paisaje pueden afectar negativamente a las aves características de espacios abiertos muchas de las cuales tienen interés en conservación (**Figura 2**). Algunas de ellas evitan o aparecen en bajas densidades en paisajes fragmentados por hábitats forestales, sugiriendo que los efectos de las reforestaciones pueden ser mayores que aquellos simplemente debidos a la pérdida de superficie de hábitats abiertos La literatura científica presenta numerosas evidencias del aumento del riesgo de depredación sobre los nidos y puestas de aves en zonas agrícolas en las que se realizan plantaciones forestales (Paton, 1994; Andren, 1995; Major & Kendal, 1996; Bayne & Hobson, 1997; Donovan *et al.*, 1997; Hartley & Hunter, 1998; Batáry & Báldi, 2004; Reino *et al.*, 2010). Las plantaciones forestales actúan como fuente de depredadores generalistas de varios tipos, incluyendo roedores, lagomorfos, gatos y perros asilvestrados y córvidos Estos depredadores generalistas normalmente tienen muy bajas densidades en hábitats desarbolados, pero prosperan en paisajes con cultivos en mosaico y plantaciones forestales exhibiendo un comportamiento explorador Particularmente, la depredación por córvidos aumenta en paisajes humanizados donde alcanzan altas densidades (Jokimaki, Huhta & Jokimäki, 2000; Chiron & Julliard, 2007, 2013; Newson *et al.*, 2010). La presencia o la ausencia de urracas (*Pica pica*) se ha mostrado en diferentes trabajos como factor clave que influye en los valores de las tasas de depredación de nidos de especies de zonas abiertas que nidifican en el suelo (Castilla *et al.*, 2007; Díaz-Ruiz *et al.*, 2010; Suvorov *et al.*, 2012; Chiron & Julliard, 2013). Trabajos previos sobre depredación de nidos de aves en hábitats abiertos adyacentes a plantaciones forestales o fragmentos forestales naturales en paisajes agrícolas mediterráneos muestran valores de las tasas de depredación entre el 13,4% y el 50% (Santos & Tellería, 1992; Castilla *et al.*, 2007; Reino *et al.*, 2010). Las urracas son atraídas fuertemente por los árboles aislados en paisajes desarbolados para nidificar y este fenómeno es muy notable en las plantaciones forestales pequeñas y aisladas en campos de cultivo reforestados.

Por otro lado, independientemente de los beneficios sociales y ambientales señalados, las plantaciones forestales pueden suponer un nuevo hábitat forestal en regiones deforestadas, ya que aumentan la heterogeneidad estructural del paisaje y favorecen a las aves forestales generalistas o ecotónicas (Santos *et al.*, 2006; Vallecillo *et al.*, 2007; Rey Benayas, Galván & Carrascal, 2010).

Las plantaciones forestales y el paisaje en el Campo de Montiel

En España central, muchos propietarios de tierras agrícolas se han acogido a las ayudas para la reforestación de la PAC desde 1993, habiéndose reforestado 111.356 ha en Castilla La Mancha correspondiendo 38.135 ha a la provincia de Ciudad Real (Junta de Castilla La Mancha, 2013). Se prevé que esta superficie reforestada aumente de una forma notable en un futuro inmediato debido, de nuevo, a la PAC. Como consecuencia de ésta, España tuvo que eliminar 175.000 ha de viñedos en el periodo 2008-2011. En la provincia de Ciudad Real, donde se extiende el mayor viñedo del mundo, se extirparon 24.347 ha de viñedo en ese periodo (Junta de Castilla La Mancha, 2013). Los propietarios de las tierras donde se han extirpado las viñas dispondrán de ayudas para reforestarlas. Por estas razones, en algunas zonas como el Campo de Montiel (Ciudad Real), se han producido y se están produciendo en la actualidad cambios acusados en el uso del suelo, cambiando la configuración del paisaje.

El paisaje del Campo de Montiel, de clima mediterráneo continental seco, está caracterizado por ser un mosaico agrícola y forestal, cuya superficie está dominada por cultivos leñosos de olivar (22%) y viñedo (18%) y cultivos herbáceos de cereal (24%), principalmente. Incluye también escasas manchas de vegetación natural y semi-natural, sobre todo de encinas, y plantaciones forestales (**Fotos 1 y 2)**. Estas plantaciones forestales están dominadas por *Pinus halepensis* (**Fotos 3-6**). Son jóvenes (<20 años) y de superficie pequeña (superficie media de 4,1 ha con muchas menores a 1 ha) debido al régimen de propiedad de la tierra en la zona, muy dividido en pequeñas parcelas (**Foto 7**). A diferencia de lo comentado en el apartado anterior, estas plantaciones no provocan un proceso de fragmentación

Fotos 1 y 2. Diferentes paisajes agrícolas con plantaciones forestales jóvenes en el Campo de Montiel. La imagen superior (**Foto 1**) muestra una plantación forestal en un paisaje heterogéneo con viñedos y olivares. En la imagen inferior (**Foto 2**) aparece una plantación forestal en un paisaje desarbolado dominado por cultivos herbáceos y pastos (Autor: Juan S. Sánchez Oliver).

Fotos 3-6. Plantaciones forestales jóvenes dominadas por pinos y, en algunos casos, acompañadas de otras especies como encinas o retamas (Autor: Juan S. Sánchez Oliver).

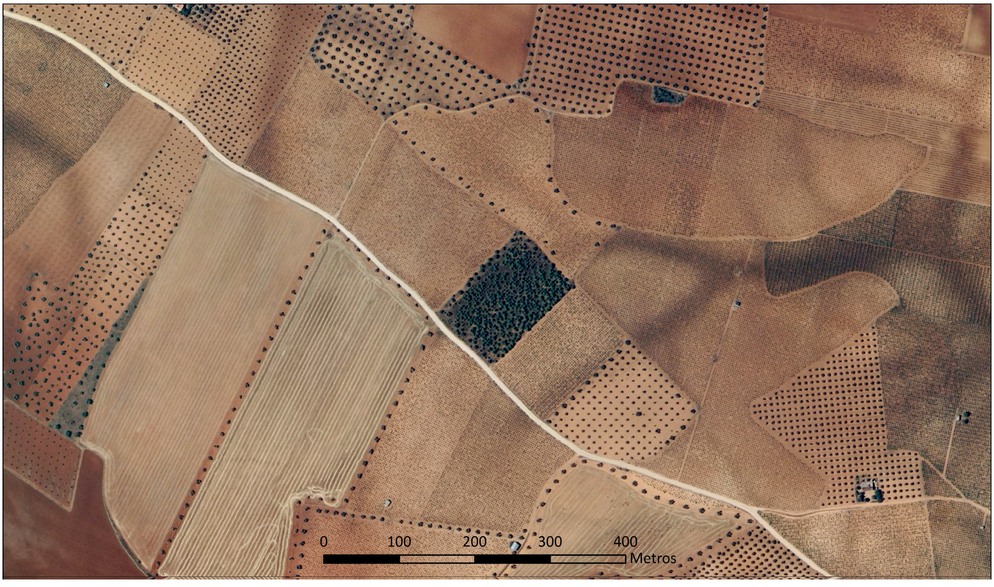

Foto 7. Mosaico agrícola compuesto por pequeñas parcelas de cultivos herbáceos, viñedos, olivares y pastos. En el centro y de color notablemente más oscuro aparece una plantación forestal joven (Autor: Juan S. Sánchez Oliver).

del hábitat agrícola en esta zona ya que debido a su pequeña superficie no rompen la continuidad de hábitats amplios y homogéneos (**Figura 4**). En este paisaje dichas plantaciones aumentan la heterogeneidad espacial y constituyen pequeñas islas arboladas en un mosaico agrícola. Estas islas de nuevo hábitat pueden ser un medio adecuado para muchas especies propias de la región, a la vez que una amenaza para especies más sensibles a alteraciones del ecosistema agrícola y, por tanto, con especial interés en conservación. En consecuencia, es interesante conocer cómo (1) las características de las plantaciones forestales, (2) las prácticas de gestión y manejo a las que son sometidas y (3) los usos del suelo en el paisaje donde son introducidas influyen en (a) la colonización de estos nuevos hábitats por la avifauna forestal regional y (b) la composición y riqueza de especies de las comunidades de aves características de ambientes agrícolas.

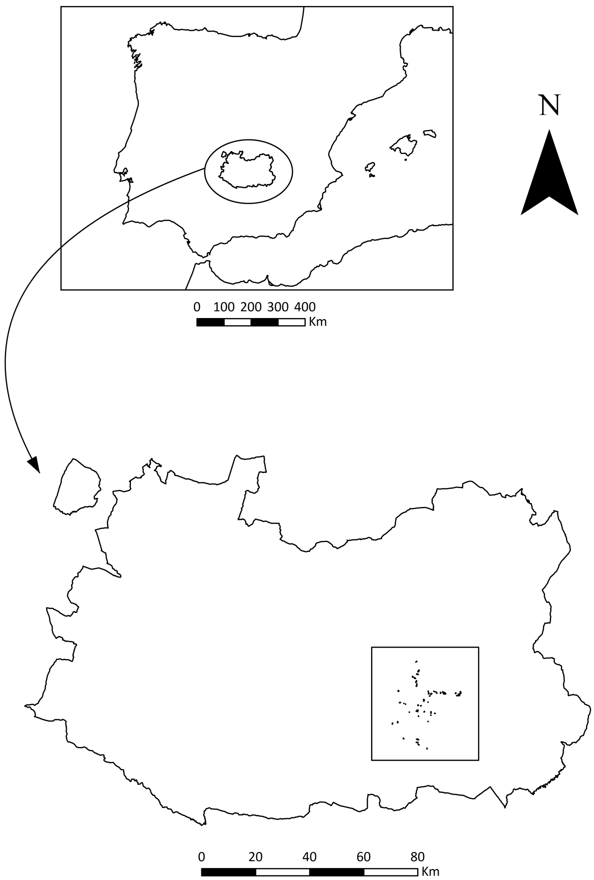

Figura 4. Localización de las plantaciones forestales en tierras agrícolas estudiadas en el Campo de Montiel (provincia de Ciudad Real).

Objetivos e hipótesis

El objetivo general de esta Tesis Doctoral es investigar los efectos de las plantaciones forestales jóvenes en campos agrícolas mediterráneos sobre las comunidades de aves. Estos efectos se estudiarán tanto en las especies que colonizan estos hábitats forestales nuevos como en las especies características de los hábitats agrícolas circundantes. Nuestra hipótesis general parte de la idea de que las plantaciones forestales en zonas agrícolas, debido tanto a sus características de vegetación como a las del paisaje que las circunda, afectarán a diversos procesos que repercutirán y contribuirán a la estructura de las comunidades de aves colonizadoras de estos nuevos hábitats y de las comunidades de aves características de los medios agrícolas. El principal fin de investigación orientada que se persigue es proporcionar propuestas de mejora de la gestión de las plantaciones forestales jóvenes en medios agrícolas.

Los objetivos específicos son los siguientes:

1. Caracterizar la composición y diversidad de la avifauna que coloniza las plantaciones forestales y establecer sus relaciones con las características (1) del hábitat local (tamaño, estructura de la vegetación) y (2) del paisaje circundante.

2. Comprobar la influencia relativa de los procesos neutros determinantes de la distribución (por ejemplo, el muestreo aleatorio o "*random sampling*"), de las preferencias de hábitat y otros rasgos autoecológicos en la presencia de las especies de aves en las plantaciones forestales jóvenes.

3. Evaluar el efecto de las plantaciones forestales jóvenes sobre la avifauna de los medios agrícolas circundantes.

4. Evaluar experimentalmente el papel de las plantaciones forestales en la depredación de nidos de aves, tanto dentro de ellas como en los hábitats agrícolas que las rodean.

Esta Memoria de Tesis Doctoral comprende siete capítulos. Se inicia con este capítulo que constituye la Introducción general (**Capítulo 1**). Los **capítulos 2-5** reproducen exactamente el contenido de los artículos enviados a diferentes revistas científicas internacionales. Como estos artículos se presentan en inglés, en cada capítulo-artículo hemos añadimos un resumen en español. Son los siguientes:

Capítulo 2: Efectos diferenciales del hábitat local y de las características del paisaje en la comunidad de aves de las plantaciones forestales promovidas por la Política Agraria Comunitaria. Este capítulo aborda el objetivo 1 y reproduce íntegramente el texto del artículo: Sánchez-Oliver J.S., Rey Benayas J.M. & Carrascal L.M. (2013) Differential effects of local habitat and landscape characteristics on bird communities in Mediterranean afforestations motivated by the EU Common Agrarian Policy. *European Journal of Wildlife Research*, DOI: 10.1007/s10344-013-0759-y.

Capítulo 3: Los efectos del muestreo aleatorio predicen la aparición de aves en reforestaciones de campos de cultivo mediterráneos. Este capítulo aborda el objetivo 2 y reproduce íntegramente el texto del artículo: Carrascal, L.M., Galván, I., Sánchez-Oliver, J.S. & Rey Benayas, J.M. (2013). Random sampling effects predict bird occurrence in Mediterranean cropland afforestations. *Ecological Research.* En revisión.

Capítulo 4: Repulsión de las aves de zonas agrícolas por las plantaciones forestales jóvenes en campos de cultivo mediterráneos. Este capítulo aborda el objetivo 3 y reproduce íntegramente el texto del artículo: Sánchez-Oliver, J.S., Rey Benayas, J.M. & Carrascal, L.M. (2013). Repellence of open-farmland birds by young afforestation of Mediterranean cropland. En revisión.

Capítulo 5: El hábitat local y el paisaje influyen en la alta depredación de nidos de aves en campos de cultivo mediterráneos reforestados. Este capítulo aborda el objetivo 4 y reproduce íntegramente el texto del artículo: Sánchez-Oliver, J.S., Rey Benayas, J.M. & Carrascal, L.M. (2013). Local habitat and landscape influence high predation of bird nests on afforested Mediterranean cropland. En revisión.

A continuación, se desarrolla una Discusión general que integra los resultados de toda la investigación (**Capítulo 6**). Para terminar, se presentan las Conclusiones más relevantes (**Capítulo 7**).

Para concluir esta Introducción general, en la **Tabla 1** presentamos una síntesis de los contenidos de esta Tesis doctoral.

Tabla 1. Resumen de los capítulos de los que consta la presente memoria de Tesis doctoral con los objetivos abordados, la comunidad de aves en la que se realizó el estudio, los métodos de censo o experimentales utilizados y los análisis estadísticos realizados.

Capítulos	**Objetivo**	**Comunidad de estudio**	**Método**	**Análisis estadísticos principales**
Capítulo 1	Introducción general			
Capítulo 2	Efecto de las características de las plantaciones y del paisaje en la colonización	Aves de las plantaciones forestales	Estaciones visuales y de escucha	*PLRS* *AIC*
Capítulo 3	Muestreo aleatorio, preferencias de hábitat y rasgos autoecológicos	Aves de las plantaciones forestales	Estaciones visuales y de escucha	*Generalized Linear Model* con binomial negativa y función log-link *AIC*
Capítulo 4	Efecto de las características de las plantaciones y del tipo de hábitat agrícola en la presencia	Aves de medios agrícolas abiertos	Transectos desde el borde de las plantaciones	*General Linear Model* *Generalized Linear Model* *PERMANOVA*
Capítulo 5	Efectos del hábitat local y del paisaje en las tasas de depredación	Aves de las plantaciones forestales y de medio agrícolas	Experimentos de depredación con nidos artificiales	*Generalized Linear Model* Modelo logístico binomial
Capítulo 6	Discusión general			
Capítulo 7	Conclusiones			

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Foto: Carbonero común (*Parus major*)

Autor: Francisco M. Silva Callejón

Capítulo 2

Efectos diferenciales del hábitat local y de las características del paisaje en la comunidad de aves de las plantaciones forestales promovidas por la Política Agraria Comunitaria

Este capítulo reproduce íntegramente el texto del siguiente manuscrito:

Sánchez-Oliver J.S., Rey Benayas J.M. & Carrascal L.M. (2013) Differential effects of local habitat and landscape characteristics on bird communities in Mediterranean afforestations motivated by the EU Common Agrarian Policy. *European Journal of Wildlife Research,* DOI: 10.1007/s10344-013-0759-y.

Resumen

Investigamos los efectos de la estructura del hábitat local y las características del paisaje circundante (proporción de tipos de usos del suelo y conectividad) en la riqueza local de especies y la composición de las comunidades de aves que habitan el interior de plantaciones forestales jóvenes en campos de cultivo del centro de España, las cuales fueron motivadas por la Política Agraria Comunitaria. La variación de la densidad de especies (número de especies en 0,78 ha) entre las plantaciones forestales mostró asociaciones ambientales diferentes entre estaciones: el hábitat local fue más importante que las características del paisaje durante el invierno, mientras que tuvieron importancia similar durante la primavera. La densidad de especies aumentó con el desarrollo del estrato arbóreo en invierno, y con la presencia de zonas urbanas alrededor de las plantaciones y la cobertura del estrato herbáceo de las mismas en el periodo reproductor. Identificamos 15 especies que exhibieron abundancias relativamente altas en hábitats forestales de la región mesomediterránea del centro de España, pero que estaban ausentes en las plantaciones forestales estudiadas en ambas estaciones. Estas plantaciones resultaron ser un hábitat atractivo para especies que explotan medios urbanos pero un hábitat desfavorable para el conjunto regional de especies forestales, excepto para las especies generalistas forestales. La composición del conjunto de especies estuvo más relacionada con la estructura del hábitat local que con las características del paisaje alrededor de las plantaciones, siendo este patrón bastante similar en invierno y primavera. Los efectos tan diferentes del hábitat local y las características del paisaje en la comunidad de aves hacen difícil sugerir prácticas de gestión con efectos positivos para todas las especies de aves durante todo el año. Concluimos que el pequeño tamaño y la poca madurez de las plantaciones forestales promovidas por las actuales ayudas de la PAC para reforestar campos de cultivo con pinos en la región Mediterránea, no contribuyen a aumentar los valores de diversidad de aves.

Palabras clave Ensamblaje de aves – Especies forestales generalistas – Usos del suelo – Variación estacional – Densidad de especies – Plantación forestal

Differential effects of local habitat and landscape characteristics on bird communities in Mediterranean afforestations motivated by the EU Common Agrarian Policy

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Abstract

We investigated the effects of local habitat structure and surrounding landscape characteristics (proportion of land use types and connectedness) on species density and composition of bird communities inhabiting the interior of young tree plantations on former cropland in central Spain, which were motivated by the Common Agrarian Policy. Variation of species density (number of species per 0.78 ha) among tree plantations showed different environmental associations across seasons: local habitat was more important than landscape characteristics during winter, while they were similarly important during spring. Species density increased with the development of the tree layer in winter, and with the presence of urban areas around tree plantations and cover of the herbaceous layer within them in the breeding season. We identified 15 species that exhibit high relative abundance in woodland habitats within the Mesomediterranean region of Central Spain that were absent in both seasons in the studied tree plantations, which were an attractive habitat for urban exploiter species but an unfavourable habitat for the regional forest species pool except for forest generalist species. Composition of bird assemblages was more related to local habitat structure than to landscape characteristics around tree plantations, and was rather similar in winter and spring seasons. The very different effects of local habitat and landscape characteristics on bird communities make difficult suggesting management practices with positive effects for all avifauna species during the entire year. We conclude that the small size and low maturity of the studied tree plantations do not contribute to enhancing the bird diversity value of current CAP aids to afforest former cropland with pines in the Mediterranean region.

Keywords Bird assemblages – Forest generalist species – Land use – Seasonal variation – Species density – Tree plantation

Introduction

Features of animal assemblages respond to the characteristics of both the local habitat and the landscape that surround such habitat, and these two sets of characteristics can interact affecting species composition and abundance (Piha *et al.*, 2007; Geiger *et al.*, 2010; Wretenberg, Pärt & Berg, 2010; Fischer *et al.*, 2011; Moreno-Mateos *et al.*, 2011). On the other hand, human activities may profoundly modify land cover and vegetation structure at both levels and, consequently, affect the composition and abundance of local communities (Blondel, 1990; Heikkinen, Luoto & Virkkala, 2004).

Large tracts of cropland and pastureland have been reforested in the world in recent decades by tree plantations or by secondary succession. Seven per cent of the forest land is tree plantations at present and their annual rate is growing as compared to afforestation by secondary succession (FAO, 2011; Rey Benayas & Bullock, 2012). These tree plantations have noticeable effects on both the abiotic environment and biological communities (Poschlod, Bakker & Kahmen, 2005; Gómez-Aparicio *et al.*, 2009; Munro *et al.*, 2009; Bremer & Farley, 2010), particularly on birds that are a taxonomic group of high indicator value (Santos *et al.*, 2006; Felton *et al.*, 2010; Lindenmayer *et al.*, 2010; Rey Benayas, Galván & Carrascal, 2010). In the European Union, the Common Agrarian Policy (CAP) has favoured the transformation of farmland into tree plantations since 1993 by means of a scheme of aid for forestry measures in agriculture (EEC Council Regulation No. 2080/92), which has resulted on the afforestation of ca. > 8 million ha to date (European Commission, 2013a, 2013b). This afforestation program pursues both societal and environmental benefits, including control of erosion, prevention of desertification, regulation of the water regime, and increasing the fixation rate of carbon dioxide. The amount of afforested farmland will likely increase in a near future in some European regions due to subsidies to vineyard extirpation (e.g. 93,600 ha were extirpated in Spain in the 2008-2011 period of which 73.1 % belonged to La Mancha; Spanish Agrarian Guarantee Fund, 2012) together with subsidies to afforestation of former vineyards, which aim to ensure EU wine production matches demand and eliminate wasteful public intervention in EU wine markets (Regulation (EC) 479/2008).

Cropland afforestations in southern Europe are mostly based on coniferous species such as *Pinus halepensis* and *P. pinaster*. Afforested fields usually form an archipelago of man-made woodland habitat in the dominant agricultural matrix. These plantations may adversely affect open habitat species that are of conservation concern in Europe, including birds, by replacing high quality steppe habitat and increasing risk of predation (Díaz *et al.*, 1998; Cresswell, 2008; Reino *et al.*, 2010). However, they may offer opportunities to woodland birds, providing suitable habitats for generalist species (Rey Benayas *et al.*, 2010). On the other hand, agricultural land abandonment and active afforestation should not be assumed to always benefit conservation, as it has been shown for birds of different biogeographic origin in agricultural lands of the Mediterranean region (increase in diversity with successional stage for Eurosiberian birds but not for Mediterranean species; Suárez-Seoane, Osborne & Baudry, 2002). Species-area relationships for bird communities in natural forests and pine plantations of Spain have been previously studied in detail (e.g. Díaz *et al.*, 1998; Santos *et al.*, 2006), demonstrating a very tight relationship between the area of forest islands and species richness. Nevertheless, little is known about how local species richness at standardized area units (i.e., species density) is affected by the surrounding landscape while taking into account habitat characteristics of the focal tree plantation.

In this study we aim to assess the wintering and breeding bird communities in young tree plantations (<20 years old) motivated by the CAP that are embedded in Mediterranean agricultural landscapes of Central Spain. These plantations are located at the south-western limit of the Palaearctic, a region with impoverished woodland avifauna dominated by species of Mediterranean origin and woodland generalists (Monkkonen, 1994; Tellería & Santos, 1994; Carrascal & Díaz, 2003), and a strong seasonality in abiotic conditions and productivity that imposes widely different ecological scenarios throughout the year on the communities living in them (Newton, 2007). They are established in small patches over a predominantly treeless landscape dominated by herbaceous or woody crops, where large mature forests of holm oaks that may serve as sources of woodland bird species are very scarce. Therefore, the avifauna in the plantations should be highly influenced by that inhabiting the surrounding landscape. This biogeographic scenario combined with the current CAP subsidies for afforestation on former arable land allow us testing the importance of local habitat characteristics and larger-scale features (e.g. the land cover surrounding the tree plantations) on bird assemblages. Moreover, the analysis of the responses of birds that colonize the interior of these afforestations in two contrasting seasons may proportionate insights about the temporal generality of their effects and suggest management practices that favour the implementation of friendly afforestation projects for woodland avifauna within deforested landscapes of the Mediterranean region on a seasonal basis.

Methods

Study area

Field work was conducted in tree plantations located in Campo de Montiel (La Mancha, situated in the southern Spanish plateau). The study area is ca. 440 km^2 within UTM coordinates x_1 4305423, x_2 4272951, y_1 458025 and y_2 483525 (zone 30S; **Fig. in Appendix 1**). Altitude ranges between 690 and 793 m a.s.l. The climate is continental Mediterranean with dry and hot summers and cold winters. Mean annual temperature and total annual precipitation in the area during the last 30 years were 13.7 °C and 390 mm, respectively (retrieved from http://www.aemet.es/). These figures were 16.6 °C and 359.9 mm in 2011, when our bird surveys took place.

The area is a representative mosaic of different crops and semi-natural or introduced woody vegetation that is characteristic of large areas in Mediterranean landscapes. Croplands were mostly occupied by herbaceous crops (wheat and barley), harvested once a year in June, and permanent woody crops (olive trees — three to five meters high, and vineyards — 1 m high). Natural vegetation mostly consisted of holm oak *Quercus rotundifolia L.* woodland and riparian forests that have been mostly extirpated from this region. Until 1992, woodland cover was restricted to open holm oak patches, usually grazed by sheep and goats. However, as in many other Mediterranean landscapes, the agricultural land is subjected to intensive management (e.g., irrigation of vineyards and olive groves) and land use change. A major result of land use change is the abandonment of herbaceous cropland and vineyard extirpation and their afforestation with the native Aleppo pine *Pinus halepensis* Mill. alone or mixed with holm oak and *Retama sphaerocarpa* (L.) Boiss., which has increased forest land in the last 20 years. These tree plantations are noticeably dominated by pines as they establish better and grow faster than the other planted species. Thus, height and diameter at breast height (dbh) of dominant pines are surrogates of the age of tree plantations.

Bird survey

First, all young forest plantations in the study area were identified using both orto-photos (Geographic Information System of Farming Land, 2010; hereafter SigPac) and Google Earth®, and were later verified in the field. We found 99 plantations that were established in 1992 or later. Next, we selected the plantations to be surveyed for birds, excluding those smaller than 1 ha: 61 forest plantations with a mean area of 4.82 ha (sd = 5.61; larger plantation = 36.5 ha). Average spacing distance between studied plantations was 11.7 km (see Fig. 1 in Appendix 1). Pruning and thinning are the management practices performed on these plantations that modify their vegetation structure; 26 of our surveyed plantations were pruned, 16 of which were also thinned.

Species abundance and density were quantified by means of circular plot censuses that were carried out in winter (January and February) and spring (April and May) 2011, to study wintering and breeding bird communities, respectively. Every tree plantation was represented by one circular plot located at the centre of the plantation. Census method consisted of point-counts (Bibby *et al.*, 2000), 10 min long each, recording all birds detected visually and/or acoustically within the 50-m radius plot (0.78 ha). We noted the presence of every bird species during the 10 min except if individuals were over-flying the plot. Two censuses of each plot were carried out in each season, one in the morning between sunrise and 3 h later and one in the evening 2 h before sunset. The relative abundance of each species and local species density (i.e., number of species per 0.78 ha) were estimated using the average of the two censuses in each season. The small area covered by the plots, and the relatively long time devoted to bird counts (accumulated census time of 20 min in each season), maximizes the detection probability of species within the area of 0.78 ha and, thus, the accurate estimations of local species density and abundance (Shiu & Lee, 2003). This time invested in bird census (25.6 min ha^{-1}) is considerably longer than that used in previous studies recording species richness in woodland islands (e.g., 10.2 min^{-1} in pine plantations sampled by Díaz *et al.*, 1998). Otherwise, our purpose was not to exhaustively characterize the avifauna of each plantation, but to analyze the variation of local species density in the interior of this novel habitat of an archipelago of young and small afforestations that punctuates the agricultural landscape. All censuses were conducted by the same well trained field ornithologist (JS S-O) on windless and rainless days.

To have a reference of the avifauna that potentially can colonize the studied plantations, we used the habitat breadth of the bird species in 15 main habitat categories as well as their relative abundance in woodlands within the Mesomediterranean region of Central Spain obtained from Carrascal & Palomino (2008).

Local habitat and landscape variables

We characterized two sets of variables related to tree plantations, namely (1) local habitat variables, which included vegetation in the bird census plots and area of the plantations, and (2) landscape variables, which included tree plantation connectivity and land use around plantations.

1. Vegetation structure and composition of main plant species at each surveyed forest plantation were measured in 25-m radius plot and a 10-m radius plot that coincided with the centres of the bird census plots. This sampling was carried out before the spring bird census. In the 25-m radius plots, we directly counted or estimated by eye, after previous training, the following structural features of the vegetation: percentage cover of chamaephytes, shrubs and trees, average height of chamaephytes, shrubs and trees, and number of trunks <5, 5-10, 10–20, 20–40 and >40cm in dbh. In the 10-m radius plots, we estimated percentage cover of herbs and bare soil and measured the average height of the herb layer. All vegetation measurements (**Table 1**) were carried out by the same observer (JS S-O) to avoid inter-personal bias.

2. Land use types were identified by means of land use layers taken from SigPac (see source above). They were analyzed with ArcGIS 10.0 in 1-km buffer-rings from the center of each forest plantation; on each buffer-ring, the percentage of area occupied by each land use type was obtained, resulting in the figures shown in **Table 1**. Finally, for a target plantation, structural connectedness was measured as the average distance of the three closest plantations or natural woodland patches weighted by the area of such plantations or woodland patches (**Table 1**).

Statistical analyses

The effects of pruning on the development of the tree layer was tested by means of a MANOVA on percentage of tree cover, height of the tree layer, dbh, and number of trunks > 5 cm.

The relationships of species density and species composition with local habitat and landscape predictor variables were separately analysed for the winter and the breeding season by means of Partial Least Squares Regressions (hereafter PLSR; Abdi, 2007). Sample units for these analyses were the 61 census plots in the tree plantations. Results obtained with PLSR are similar to those from conventional multiple regression techniques; however, PLSR allows for the simultaneous analysis of multiple response variables, and it is extremely robust to the effects of low sample size (i.e. overfitting) and high degree of correlation between predictor variables (i.e. severe multi-collinearity) (Carrascal, Galván & Gordo, 2009). PLSR establishes associations between the response variables and factors extracted from the predictor variables that maximize the explained variance in the response variables. These factors are defined as linear combinations of predictors, so the original multidimensionality is reduced to a lower number of orthogonal factors to detect structure in the relationships between predictor variables and between these factors and the response variables. The relative contribution of each predictor to the extracted factors was calculated by means of the square of predictor weights. The PLSR components regarding species composition were obtained based on the abundance of those species with >0.1 birds/census plot; the abundance of 12 species in winter and 17 species in spring defined the response variables that were summarized in composition components by means of the linear combination of the species' abundances. Only those components significant after a ten-fold validation procedure were retained (StatSoft, 2011).

All statistical analyses were conducted in Statistica 10 (StatSoft, 2011).

Results

Tree plantation and landscape characteristics

There was a broad variation in the local habitat variables of the studied tree plantations (**Table 1**). Overall, the number of pines >5 cm dbh was not too large but there were a lot of small trees when considering the average trunk diameter of pines. Pruning enhanced the development of the tree layer according to a MANOVA (Wilk's λ = 0.752, p = 0.003, n = 61).

There was also a considerable variation in the landscape characteristics around the tree plantations in an area mainly dominated by dry herbaceous cropland, olive tree groves and vineyards (**Table 1**).

Table 1. Mean, standard deviation (sd) and range (min / max) of the local habitat and landscape variables describing the characteristics of the 61 studied tree plantations.

	Mean	SD	Range	
Local habitat				
Area of tree plantation (ha)	4.8	5.6	1.0	36.5
Cover of the tree layer (%)	35.4	25.5	1.7	100
Average pine height (m)	3.5	1.5	0.9	7.2
Average trunk diameter of pines (dbh cm)	11.4	5.8	0	33.2
# of pine trunks larger than 5 cm dbh / 0.2 ha	70.5	50.7	0	185
Cover of the shrub layer (%)	4.7	8.8	0	46.2
Average height of the shrub layer (m)	1.2	1.1	0	3.3
Cover of the herbaceous layer (%)	54.3	40	0	100
Average height of the herbaceous layer (m)	0.4	0.3	0	1.1
Landscape around plantations				
Average distance to other woodlands (m)	739.7	621.7	14	2506
Streams, rivers and lagoons (% cover)	0.7	1.1	0	4.1
Roads and rural tracks (% cover)	6.4	5.2	0	31.1
Woodlands (% cover)	4.2	4.7	0.1	25.2
Fruit groves (% cover)	1.1	1.3	0	5.4
Waste lands (% cover)	6.8	4.4	0	14.8
Olive groves (% cover)	21.9	23.7	0	94.7
Pastures with scattered trees (% cover)	0.4	1.6	0	9.4
Scrubland (% cover)	10.0	7.4	0	29.5
Pastures (% cover)	1.1	3.2	0	19.1
Dry herbaceous cropland (% cover)	18.2	9.2	0	40.8
Vineyards (% cover)	20.9	13.7	0	49.2
Vineyards with olive trees (% cover)	5.1	8.5	0	32.3
Dried fruit orchards (% cover)	0.6	2.4	0	16.9
Urban areas and scattered buildings (% cover)	2.4	4.2	0	25.8

Species density

Average number of species per census plot of 0.78-ha did not significantly change between seasons (paired t test: t=0.158, df=60, p=0.875), being 4.38 species during winter time (range=0-9, SD=2.02, n=61 plots) and 4.43 species during the breeding season (range=1-10, SD=1.84). Winter and spring species density were not significantly correlated (r=0.208, p=0.109, n=61).

One significant component (p<<0.001) was obtained in each PLSR analysis of species density in the 61 studied tree plantations using all local habitat and landscape predictor variables, accounting for 31.9 % and 31.4 % of total variance in winter and breeding season species density, respectively (**Table 2**; **Fig. 1**). Environmental effects on local species density were very different in both seasons. The weights of local habitat and landscape variables were not significantly correlated in winter and spring (r=0.190, p=0.372, n=24 predictor variables), thus defining different patterns of environmental determinism on species density in both seasons.

In winter, species density mainly increased with the development of the tree layer (cover, height and trunk diameter of pines), which was associated to low development of the herbaceous and shrub layers (**Table 2; Fig. 1**). None predictor variable describing landscape characteristics around the plantations attained a |weight|>0.2. Thus, the importance of local habitat variables on winter species density was considerably higher than the importance of variables describing the landscape characteristics (calculated by means of the square of predictor weights), and was considerably higher than that expected considering the relative number of predictors in the two groups of variables (local habitat=0.86, landscape=0.14; the 'null' proportions according to the number of predictors was 0.38 for nine local habitat variables and 0.62 for 15 landscape variables).

During the breeding season, species density was positively associated with the presence of waste lands, urban areas and scattered buildings around them, and negatively related to their size, the height and cover of shrubs and the amount of area around plantations covered by woodland (mainly remaining patches of holm oak forests), fruit groves, shrubland, and dry herbaceous cropland (**Table 2; Fig. 1**). Characteristics of landscape surrounding the tree plantations were similarly important than local habitat in determining species density during the breeding season (summatory of the square of predictor weights: 0.42 for nine local habitat variables and 0.58 for 15 landscape characteristics, which were very similar to the "null" proportions of 0.38 and 0.62, respectively, according to the number of predictors).

Table 2. Results of the partial least squares regression (PLSR) models analyzing the variation in bird species density and bird species composition in 61 tree plantations during winter and the breeding season (spring) according to nine local habitat features of plantations and 15 landscape predictor variables.

	SPP DENSITY		SPP COMPOSITION	
	Winter	Spring	Winter	Spring
Local habitat				
Area of tree plantation (ha)	0.16	**-0.29**	0.02	0.01
Cover of the tree layer (%)	**0.39**	0.09	**0.42**	**0.41**
Average pine height (m)	**0.47**	0.12	**0.45**	**0.45**
Average trunk diameter of pines (dbh cm)	**0.41**	-0.03	**0.34**	**0.29**
# of pine trunks larger than 5 cm dbh	**0.41**	-0.09	**0.41**	**0.33**
Cover of the shrub layer (%)	-0.06	**-0.39**	-0.16	-0.16
Average height of the shrub layer (m)	-0.22	**-0.34**	**-0.28**	**-0.23**
Cover of the herbaceous layer (%)	-0.10	0.19	-0.08	**-0.21**
Average height of the herbaceous layer (m)	**-0.25**	-0.03	**-0.26**	**-0.20**
Landscape around plantations				
Average distance to other woodlands (m)	-0.06	0.08	0.07	0.14
Streams, rivers and lagoons (% cover)	0.08	-0.03	0.00	0.06
Roads and rural tracks (% cover)	0.17	-0.06	0.03	0.01
Woodlands (% cover)	-0.01	**-0.29**	-0.14	0.00
Fruit groves (% cover)	-0.15	**-0.34**	-0.12	-0.11
Waste lands (% cover)	0.01	**0.27**	-0.04	-0.17
Olive groves (% cover)	-0.07	0.08	-0.01	0.10
Pastures with scattered trees (% cover)	0.02	0.08	0.08	-0.04
Scrubland (% cover)	0.04	**-0.33**	-0.04	-0.04
Pastures (% cover)	0.01	0.02	0.08	-0.01
Dry herbaceous cropland (% cover)	-0.06	**-0.20**	-0.08	-0.11
Vineyards (% cover)	0.07	0.05	0.01	-0.04
Vineyards with olive trees (% cover)	0.06	0.19	0.19	**0.22**
Dried fruit orchards (% cover)	0.18	-0.13	**0.21**	0.09
Urban areas and scattered buildings (% cover)	-0.18	**0.26**	-0.16	**-0.35**

Figures shown are the predictor weights of each variable in each component (in bold those with |weights| > 0.2; this threshold was calculated according to the following equation: $[1 / \#predictors]^{0.5}$).

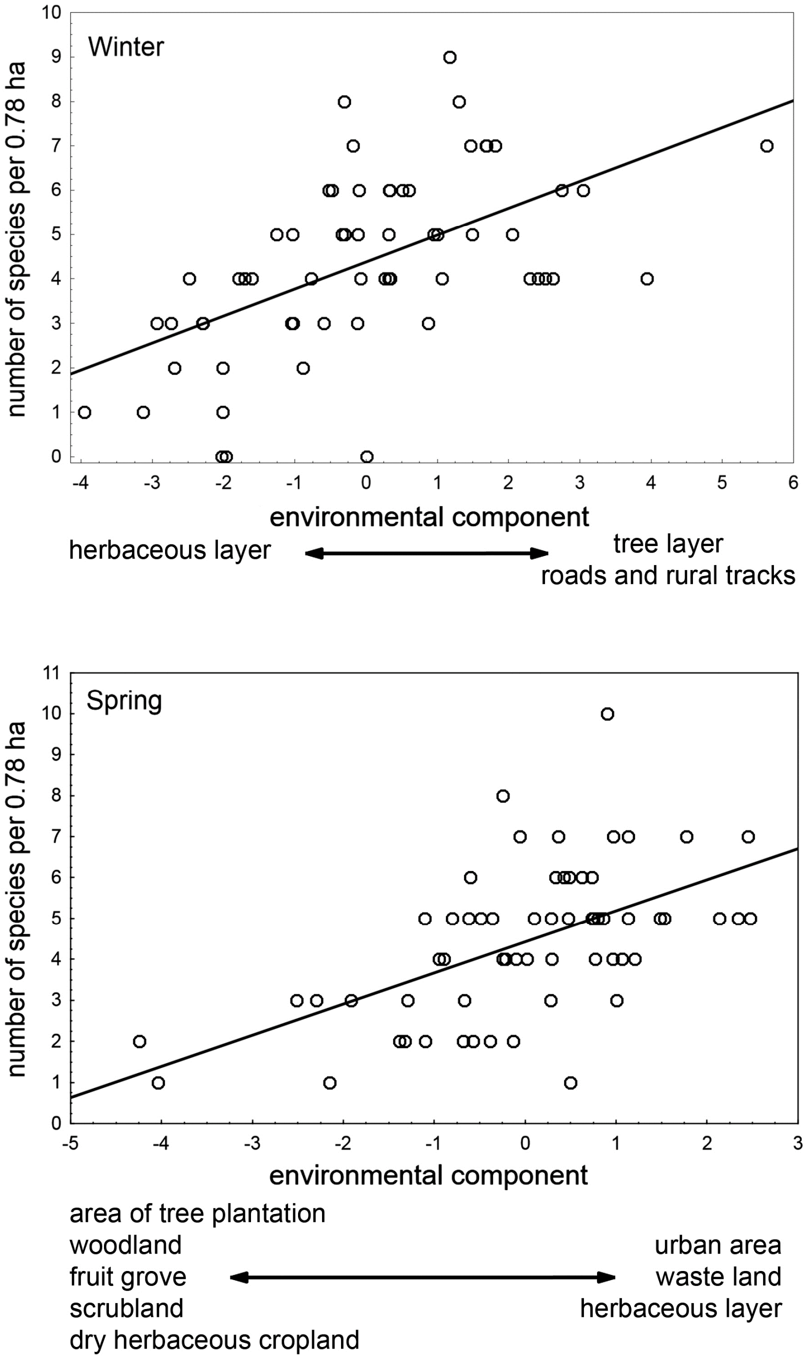

Figure 1. Relationship between (a) the species density per 0.78-ha census plot of tree plantations in the winter (top) and (b) the breeding season (down) and the multivariate gradient (first PLSR component) defined by the Partial Least Squared Regression analysis on nine local habitat and 15 landscape predictor variables.

Species composition

The avifauna was dominated by the great tit (*Parus major*), the chiffchaff (*Phylloscopus collybita*), the goldfinch (*Carduelis carduelis*), the wood pigeon (*Columba palumbus*) and the magpie (*Pica pica*) in wintertime, and by the goldfinch, the spotless starling (*Sturnus unicolor*), the wood pigeon and the magpie during the breeding season (spring) (average of more than one detected individual per census plot in both seasons; **Appendix 2**).

The following species that exhibit high relative abundance in woodland habitats within the Mesomediterranean region of Central Spain according to Carrascal & Palomino (2008) were completely absent in both seasons in the studied tree plantations: great spotted woodpecker (*Dendrocopos major*), blackbird (*Turdus merula*), nuthatch (*Sitta europaea*), short-toed treecreeper (*Certhia brachydactyla*), firecrest (*Regulus ignicapillus*), coal tit (*Periparus ater*), crested tit (*Lophophanes cristatus*), long-tailed tit (*Aegithalos caudatus*), hawfinch (*Coccothraustes coccothraustes*), blue tit (*Cyanistes caeruleus*), rock bunting (*Emberiza cia*), jay (*Garrulus glandarius*), and Eurasian hoopoe (*Upupa epops*). Similarly, other woodland species in the region such as robin (*Erithacus rubecula*) and woodchat shrike (*Lanius senator*) were very scarce in the studied plantations.

Relative abundances of species across the 61 studied tree plantations were not tightly correlated among themselves either in winter or during the breeding season, as defined by the low variance attained by the first components of the PLSRs in both seasons using the common species (those with more than 0.1 birds/plot): 7.9 % of variance in the relative abundances of 12 species in winter, and 5.7 % of variance for 17 species in spring. Nevertheless, these loose patterns of co-variation in species abundances were highly associated with the plantation characteristics, mainly local habitat in both seasons (see below): r=0.675, p<<0.001 for winter and r=0.700, p<<0.001 for the breeding season.

The main pattern of co-variation in species abundances during the winter season was the association of the chiffchaff, great tit, magpie, wood pigeon, chaffinch (*Fringilla coelebs*), and goldfinch in tree plantations with a well developed tree layer (**Fig. 2**; see predictor variable weights in Table 2; these species were selected according to absolute values of loadings >0.2 in the component of species abundances). On the other hand, there is a common pattern of increase in species abundances during the breeding season that associates the magpie, great tit, and wood pigeon in tree plantations with a tall and dense cover

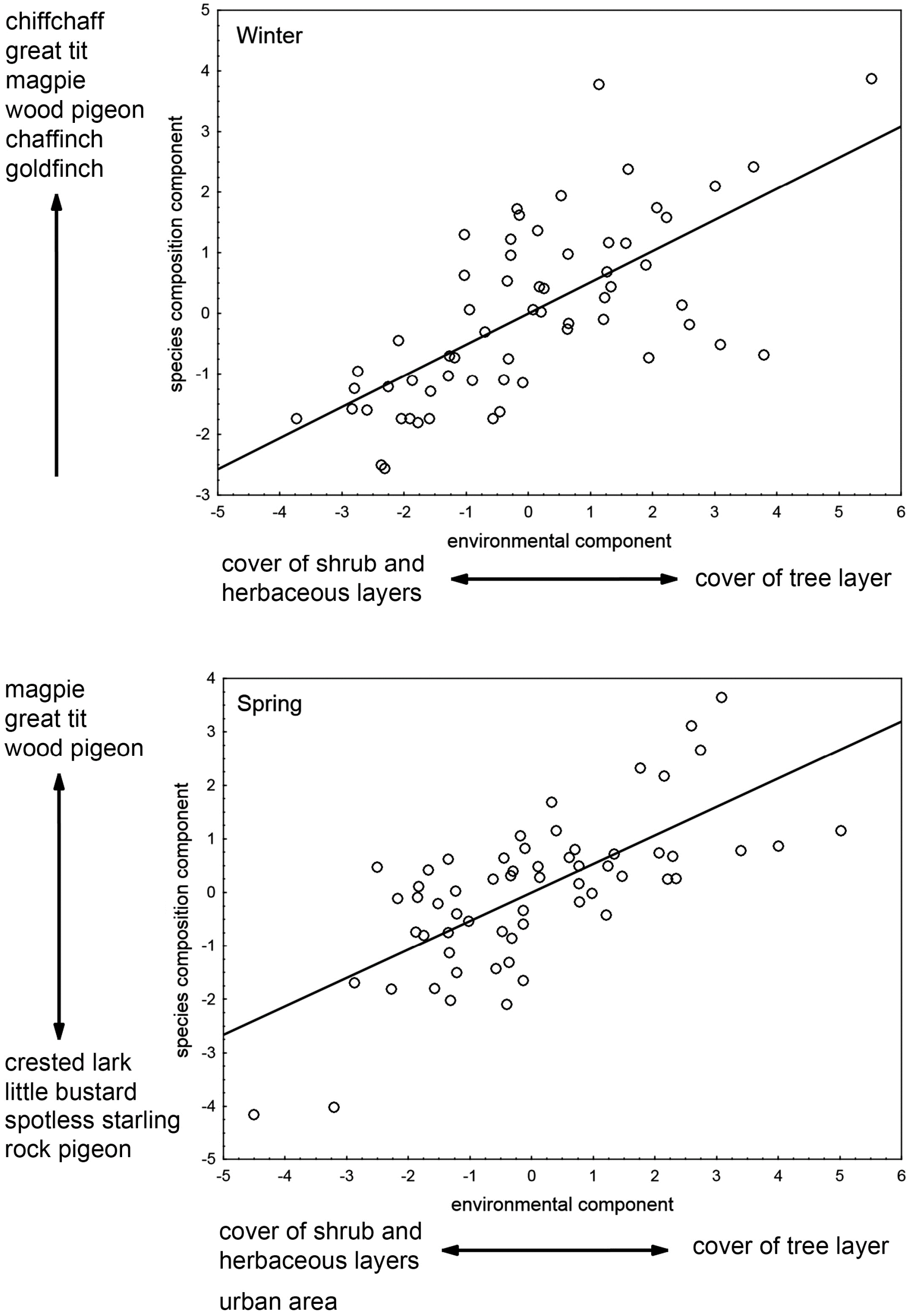

Figure 2. Relationship between (a) the species composition of tree plantations in the winter (top) and (b) the breeding season (down) and the multivariate gradient (first PLSR component) defined by the Partial Least Squared Regression analysis on nine local habitat and 15 landscape predictor variables.

of the tree layer surrounded by relatively high cover of vineyard with olive trees, as opposed to the co-variation of abundances of rock pigeon (*Columba livia*), spotless starling, little bustard (*Tetrax tetrax*) and crested lark (*Galerida cristata*) in plantations with high cover of the shrub~~s~~ and herb layers near urban areas (**Fig. 2**).

The importance of the environmental factors related to composition of bird assemblages was rather similar in winter and spring (**Table 2**), as the weights of local habitat and landscape variables were highly correlated in both seasons: r=0.921, p<<0.001, n=24 predictor variables). Moreover, the importance of local habitat variables in defining the co-variation of abundance of bird species was considerably higher than that of variables describing the landscape characteristics around tree plantations in both seasons calculated by means of the square of predictor weights (winter—local habitat=0.83, landscape=0.17; spring—local habitat=0.73, landscape=0.27; the 'null' proportions according to the number of predictors were 0.38 and 0.62, respectively).

Discussion

Overall community composition

Our results show that the local composition of bird assemblages inhabiting the interior of young Mediterranean cropland afforestations are characterized by a few common dominant species, namely magpie, wood pigeon and goldfinch in both seasons, great tit and chiffchaff in wintertime, and spotless starling in spring. These ubiquitous species are generalist birds of wooded areas, with broad geographical ranges and high population sizes in Spain (Martí & del Moral, 2003; Carrascal & Palomino, 2008). They are of little conservation concern in the European context (BirdLife International, 2004). They are also of little sensibility to habitat fragmentation as they can thrive in very small woodland patches (Díaz *et al.*, 1998; Santos, Tellería & Carbonell, 2002; Razola & Rey Benayas, 2009), such as those corresponding to the afforestations investigated in this study.

The biogeographical basis of the avifauna in this Mediterranean region, with an impoverished European forest avifauna dominated by species of early successional stages, probably limits the possibility of colonization of pure coniferous woodland species. Forest specialists of Mediterranean coniferous forests that require more mature and larger woodland patches (Díaz *et al.*, 1998; Santos *et al.*, 2006), such as the great spotted woodpecker, firecrest, crested tit, short-toed treecreeper or nuthatch, were never recorded in these plantations, thus emphasizing the low suitability of these woodlands for forest avifauna of the region. This points to the importance of the biogeographic context when designing restoration plans with afforestations in agricultural-dominant landscapes (Suárez-Seoane *et al.*, 2002), and enlightens the conflicts that can arise if single services of ecological restoration such as carbon sequestration by tree plantations are targeted without taking into account regional biodiversity (Bullock *et al.*, 2011).

Relative effects of local habitat and landscape characteristics

The influence of different sets of environmental factors, namely local habitat of tree plantations and landscape characteristics, on bird communities changed considerably between seasons, with a prominent role of local habitat variables during winter for species density, and a more balanced importance of landscape characteristics around plantations and local habitat during the breeding season. During the breeding season birds are spatially restricted to the focal place where they breed, and thus they show marked habitat preferences related to vegetation structure, which is an important attribute determining species composition of bird communities at the local scale (Hurlbert, 2004; Hinsley *et al.*, 2009). In contrast, during the winter period, birds adopt a vagabonding life-style exploring a greater variety of habitats over larger areas to track the spatial and temporal distribution of food availability (Levey & Stiles, 1992; Wiktander, Olsson & Nilsson, 2001). From this perspective, local habitat should have a greater importance in the breeding season than in the winter in influencing bird communities of tree plantations within agricultural landscapes. Nevertheless, our results do not support this prediction for species density.

The negative influence of the area of tree plantations studied here on local species density is related to the fact that the probability of recording “ubiquitous/edge” bird species in the centre of plantations decreases as plantation area increases. This result, together with the remarkable negative influence of nearby woodlands on local species density in the interior of the plantations, reinforces the idea of the low favourability of these young afforestations dominated by pines for the forest avifauna of the study region, especially if they are large.

The high importance of urban cover around the tree plantations on species density during spring points to the attractiveness of scarce woodland fragments to urban-exploiters of Central Spain (Palomino & Carrascal, 2006), such as the rock dove (*Columba livia*), collared dove (*Streptopelia decaocto*), greenfinch (*Carduelis chloris*), house sparrow (*Passer domesticus*), magpie, or spotless starling. It also emphasizes that urban development extends its impact on the surrounding habitats affecting bird communities, especially by the influence of just a few very common urban species (e.g. Findlay & Houlahan, 1997; Sauvajot *et al.*, 1998; Odell & Knight, 2001; Palomino & Carrascal, 2007). Urban and surrounding areas are a source of the ubiquitous and opportunistic nest predator magpie, and could thus entail additional conservation concern, because its overabundance around the cities could pose a deleterious effect on other bird species breeding in the plantations (e.g. Andren, 1992; Groom, 1993; Paradis *et al.*, 2000). Similarly, Lindenmayer *et al.* (2012) found that another aggressive corvid reduced bird abundance in Australian tree plantations located in an agricultural landscape.

Management of tree plantations

The results of this study show that, overall, there are difficulties in making generalizations about the environmental factors that determine bird diversity inhabiting the interior of young tree plantations in Mediterranean agricultural landscapes on a year-round basis, and thus in outlining management recommendations to make them friendlier for the avifauna. These plantations offer opportunities for a few generalist forest bird species but are not perceived as an attractive breeding habitat for most forest species in the region. Further, the youngest plantations with under-developed tree layer and presence of shrub and herbaceous layers benefit bird species that are characteristic of open farmland habitats such as the calandra lark, little bustard and rock pigeon (Rey Benayas *et al.*, 2010). As pruning of pines speeds up the development of the tree layer, a more generalized use of this practice would increase overall species density in winter and benefit forest species such as the wood pigeon, which is of interest to hunters, and insectivorous birds such as the great tit or blue tit, which have the potential of enhancing pest regulation in both tree plantations and crops around them (Jedlicka, Greenberg & Letourneau, 2011).

Conclusions

Local habitat and surrounding landscape characteristics in Mediterranean landscapes dominated by croplands had very different effects on bird communities inhabiting the interior of young afforestations in the winter and breeding seasons, which make difficult suggesting extensive management practices with positive effects for all avifauna species during the entire year. These small, monotonous plantations are an attractive habitat for urban exploiter species but an unfavourable habitat for the regional forest species pool with the exception of the forest generalist species. Therefore, the small size and low maturity of the studied tree plantations do not contribute to enhancing the bird diversity value of current CAP aids to afforest former cropland with pines in the Mediterranean region. Further monitoring of bird communities as these plantations get older is necessary to provide more robust science-based management recommendations, and test the success of the implemented recommendation (more use of tree pruning) that the results of this study hinted.

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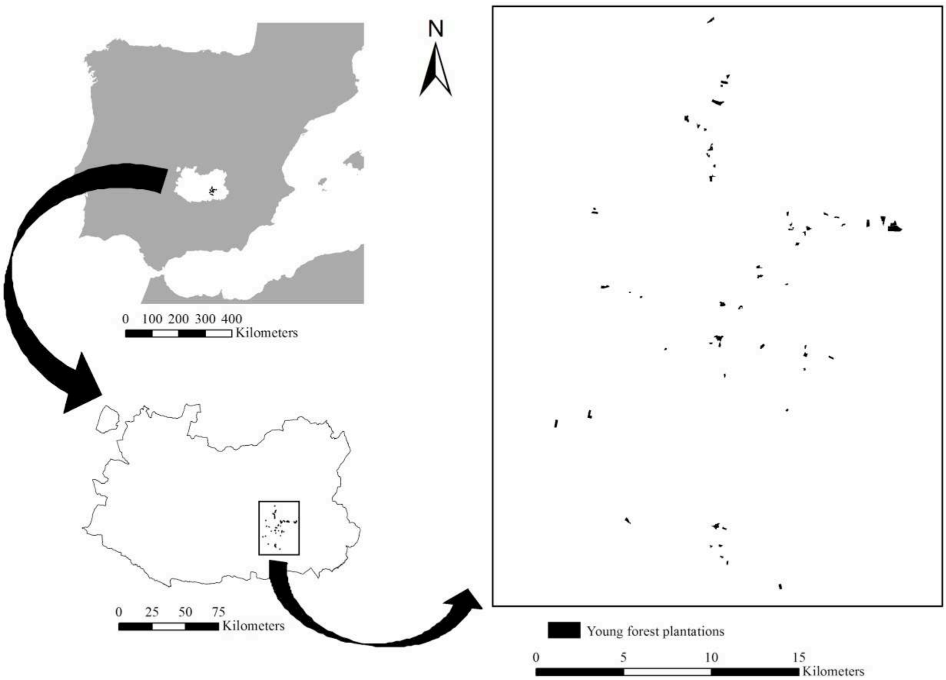
Appendix 1. Location of the study area in central Spain within the Ciudad Real province and distribution of the young forest plantations on cropland that was investigated in this study.



Appendix 2. Relative abundance of birds detected in 50-m radius circular plots (0.78 ha) in the two study seasons (--: species not detected in that season).

	Winter	Spring
Alauda arvensis	0.03	0.16
Carduelis cannabina	0.28	0.33
Carduelis carduelis	1.64	3.89
Carduelis chloris	--	0.07
Clamator glandarius	--	0.30
Columba palumbus	1.03	1.93
Columbia livia	0.02	0.15
Coturnix coturnix	--	0.02
Cyanistes caeruleus	0.16	0.03
Erithacus rubercula	0.02	--
Fringilla coelebs	1.16	0.05
Fringilla montifringilla	0.02	--
Galerida cristata	0.05	0.41
Lanius meridionalis	0.03	0.02
Lanius senator	--	0.03
Melanocorypha calandra	0.10	0.16
Miliaria calandra	--	0.26
Motacilla alba	0.03	--
Periparus ater	--	0.02
Parus major	2.70	0.52
Passer domesticus	0.82	0.33
Passer montanus	--	0.08
Petronia petronia	0.02	--
Phylloscopus collybita	2.57	0.08
Phoenicurus ochruros	0.02	--
Pica pica	0.97	1.00
Serinus serinus	0.03	0.05
Streptopelia decaocto	--	0.16
Sturnus unicolor	0.46	3.08
Sylvia atricapilla	0.03	--
Tetrax tetrax	--	0.23
Turdus merula	0.02	0.02
Turdus philomelos	0.02	--
Turdus viscivorus	0.03	--
Upupa epops	--	0.08

Foto: Herrerillo común (*Cyanistes caeruleus*)

Autor: Francisco M. Silva Callejón

Capítulo 3

Los efectos del muestreo aleatorio predicen la aparición de aves en reforestaciones de campos de cultivo mediterráneos

Este capítulo reproduce íntegramente el texto del siguiente manuscrito:

Carrascal, L.M., Galván, I., **Sánchez-Oliver, J.S.** & Rey Benayas, J.M. (2013). Random sampling effects predict bird occurrence in Mediterranean cropland afforestations. Ecological Research. En revisión.

Resumen

Parte de los cultivos abandonados en los paisajes mediterráneos están siendo sustituidos por reforestaciones dominadas por pinos. En este estudio, evaluamos simultáneamente el efecto de tres categorías de factores que pueden explicar la variación interespecífica en la ocupación de este hábitat muy fragmentado por aves, considerando los efectos del muestreo aleatorio, las preferencias de hábitat y los rasgos de historia de vida de las especies. Usamos el "experimento natural" que las plantaciones de pinos muy fragmentadas del centro de España representan debido al modelo de pequeñas propiedades. Algunas especies con marcadas preferencias por los hábitats forestales fueron muy escasas o nunca se registraron en estos hábitats noveles situados en una matriz de paisaje agrícola deforestado. La variabilidad interespecífica en la presencia de las especies fue principalmente explicada por los efectos del muestreo aleatorio, de tal modo que estuvo asociada positiva y significativamente con la extensión del área de distribución regional alrededor del área de estudio, la amplitud regional de hábitat y las tendencias poblacionales de las especies en el centro de España durante el periodo 1998-2011. También estuvo asociada positivamente con la ocupación regional de plantaciones maduras de pinos. Otros rasgos de las especies relacionados con las preferencias de hábitat (por medios forestales, agrícolas y urbanos), la talla corporal, el tamaño de puesta o su carácter migratorio no estuvieron relacionados con la presencia de las especies. Por tanto, estas pequeñas islas de pinos creadas por el hombre gracias a la Política Agraria Comunitaria (PAC), en un paisaje dominado por hábitats agrícolas mediterráneos, sólo capturan especies de aves generalistas y ampliamente distribuidas y con incrementos poblacionales recientes, no favoreciendo a especies forestales especialistas.

Palabras clave Presencia de aves – Abandono de tierras agrícolas – Preferencias de hábitat – Plantaciones de pinos – Muestreo aleatorio

Random sampling effects predict bird occurrence in Mediterranean cropland afforestations

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Abstract

Part of the abandoned cropland in Mediterranean landscapes is being subjected to afforestation dominated by pines. Here we simultaneously evaluate the effect of three categories of factors as predictors of the interspecific variation in bird habitat occupancy of fragmented afforestations, namely random sampling effects, habitat preferences and life history traits of species. We use the "natural experiment" that highly fragmented pine plantations of Central Spain represent due to the prevailing pattern of land ownership of small properties. Many species with marked habitat preferences for woodland habitats were very scarce or were never recorded in this novel habitat within a matrix of deforested agricultural landscape. Interspecific variability in occurrence was mainly explained by random sampling effects: occurrence was significantly and positively associated with the proportion of occupied 10x10 UTM km squares around the study area, regional habitat breadth, and population trend of species in the period 1998-2011. It was also positively associated with regional occupancy of mature and large pine plantations. Other predictor variables related to habitat preferences (for woodland, agricultural and urban habitats) or life history traits (migratory strategy, body mass and clutch size) were unrelated to the occurrence of species. Thus, small man-made pinewood islands funded by the Common Agrarian Policy (CAP) within a landscape dominated by Mediterranean agricultural habitats only capture

widespread and habitat generalist avian species with increasing population trends, not contributing to enhance truly woodland species.

Keywords Bird occurrence · Cropland abandonment · Habitat preferences · Pine plantations · Random sampling

Introduction

Afforestation represents a strategy to produce forest land on abandoned cropland that avoids the long time that secondary succession usually takes, particularly in the drylands of the world (Rey Benayas & Bullock, 2012). In the European Union, the Common Agrarian Policy (CAP) has favoured the transformation of farmland into tree plantations since 1992 by means of a scheme of aid for forestry measures in agriculture (EEC Council Regulation No. 2080/92), which has resulted on the afforestation of ca. > 8 million ha to date (European Commission, 2013a, 2013b). These afforested fields, which in southern Europe are mostly based on coniferous species such as *Pinus halepensis* and *P. pinaster*, usually form an archipelago of habitat patches in the dominant agricultural matrix (Izhaki, 1999; van Meijl *et al.*, 2006). These afforestations impact on biodiversity because pine plantations have higher tree cover and less structural heterogeneity than natural Mediterranean woodlands (Sirami et al. 2007; Rey Benayas et al. 2010).

Birds represent the group of vertebrates upon which these impacts have been most intensively studied, and are good indicators of the success of colonization of these afforestations because they are highly mobile animals that easily reach these novel ecosystems. It is widely known that the age, area and habitat structure of woodland islands are tightly related to species richness and bird community composition of pine plantations (e.g. Díaz *et al.*, 1998; Shochat, Abramsky & Pinshow, 2001). Nevertheless, little is known about the filtering processes determining the identity of species occupying pine plantations within the regional pool of species in Mediterranean landscapes dominated by agricultural habitats. Three major types of effects may determine bird species identity in Mediterranean cropland afforestations, namely random sampling effects, habitat preferences of species and autoecological traits related to life history. Random sampling claims that species with high density are more likely to be included in habitat fragments than scarce ones as a consequence of their abundance (Connor and McCoy 1979; Andrén 1999), and predicts that the bird community inhabiting afforested fields would represent random subsets of the regional species pool independently of any ecological process. In fact, density seems to be a good predictor of the probability of species' presence in forest fragments (e.g. Bolger, Alberts & Soule, 1991; Tellería & Santos, 1997, 1999). Additionally, species that tolerate a relatively wide range of ecological conditions are in turn more widespread and are able to occupy a large variety of habitats (Swihart *et al.*, 2003; Bohning-Gaese *et al.*, 2006; Hurlbert & White, 2007; Carrascal *et al.*, 2008). Therefore, pine plantations within a matrix of arable land would not selectively filter species according to the habitat characteristics of the plantations, but they would be occupied by those eurytopic species that are highly spread and abundant in the region.

However, pine plantations could impose a selective filter to the colonization by the species of the regional pool according to their strict habitat selection and spatial niches (e.g. preferences for pine foliage or trunks for foraging or nesting). The habitat requirements of bird species that are characteristic of the dominant agricultural environments, or of Mediterranean habitats with marked preferences for broadleaf sclerophyllous foliage or for a well developed understory (e.g. Tellería, Asensio & Díaz, 1999), contrast with structural characteristics provided by coniferous plantations with a generally poor shrub layer. Therefore, the habitat-matching hypothesis predicts that habitat structure of afforestations would constrain its colonization by the regional pool of species, favouring only those birds which exhibit a marked preference for coniferous trees and a high cover of the tree canopy (Santos *et al.*, 2006; Sirami *et al.*, 2008a; Sirami, Brotons & Martin, 2008b; Rey Benayas *et al.*, 2010). Moreover, birds may have a higher colonization success of pine plantations, in a landscape dominated by arable lands, if they are able to survive in human-altered novel habitats such as urban environments. Several studies have shown that urban-exploiter birds have larger brains than urban-avoider species (Maklakov *et al.*, 2011), probably because large-brained animals are behaviourally innovative species that have higher success and experience lower mortality when exposed to a novel environment (Sol *et al.*, 2005a, 2007, 2008).

Some autoecological traits of bird species, such as sedentariness, body mass and clutch size may also predict the colonization success of pine plantations within the matrix of arable lands. Smaller birds usually attain high local densities according to the inverse allometric relationship 'body mass–population density', a pattern more pronounced in assemblages exploiting foliage or in habitats with a high foliage volume, in which smaller bird species predominate because of ecomorphological constraints (low body mass for hovering and hanging; e.g., Tellería & Carrascal, 1994). Moreover, small body size is a predictor of establishment success across species (Cassey, 2001). Species that are sedentary and have large clutch sizes are more likely to occupy pine plantations, as these species are more likely to visit and explore the ecological opportunities provided by afforestation on a year-round basis and may attain high rates of population growth (Galván & Rey Benayas, 2011), whereas resident species tend to rely more on innovative feeding behaviours in winter when food is harder to find (Sol, Lefebvre & Rodríguez-Teijeiro, 2005b). In short, the factors that affect the success of pine plantations as a restoration strategy in Mediterranean croplands, as reflected by the capacity to hold bird species, can be divided into passive processes and active filters related to habitat requirements and preferences and autoecological traits of the bird species.

Here we evaluate interspecific differences in bird species occurrence in the novel fragmented habitat provided by small and young pine plantations of Central Spain, established over a predominantly treeless landscape dominated by herbaceous or woody cultures, where large mature forests of holm oak *Quercus rotundifolia* that may serve as sources of woodland bird species are very scarce. This habitat consists of an archipelago of young and small afforestations that punctuates the agricultural landscape, because it has been favoured by the EU CAP in the early 90s, and the size of the cropped fields is usually small (< 5 ha) due to the pattern of land ownership. The woodland avifauna of this region is impoverished and is dominated by species of Mediterranean origin and woodland generalists, as abundance of many forest birds decreases along a north-west/mesic to south-east/xeric gradient (Tellería & Santos, 1993, 1994), especially for those of European biogeographic origin (Carrascal & Díaz, 2003). The "natural experiment" associated with these plantations allows us to ascertain the relative influence of random sampling versus habitat preferences and some auto-ecological traits of the bird species in determining the occupancy of the referred novel afforested habitat. We hypothesize that bird species with a higher occurrence in afforested fields will be those with (i) broader geographical distribution, larger habitat breadth and increasing population trends in the recent years, (ii) marked preferences for woodland habitats and with high occupancy of novel habitats, and (iii) sedentary migration strategy, large clutch size and small body mass.

Materials and methods

Study area

Field work was conducted in tree plantations located in Campo de Montiel (38º 45’ 36” N, 3º 23’ 7” W; La Mancha, situated in the southern Spanish plateau; **Figure 1**). The study area spreads on 440 km^2 and altitude ranges between 690 and 793 m a.s.l. The area is included within the Mesomediterranean bioclimate region of the Iberian Peninsula (Rivas Martínez, 1981). The climate is continental Mediterranean with dry and hot summers and cold winters. Mean annual temperature and total annual precipitation in the area during the last 30 years were 13.7 °C and 390 mm, respectively (Agencia Española de Meteorología, 2012). These figures were 16.6 °C and 359.9 mm in 2011, when bird surveys took place.

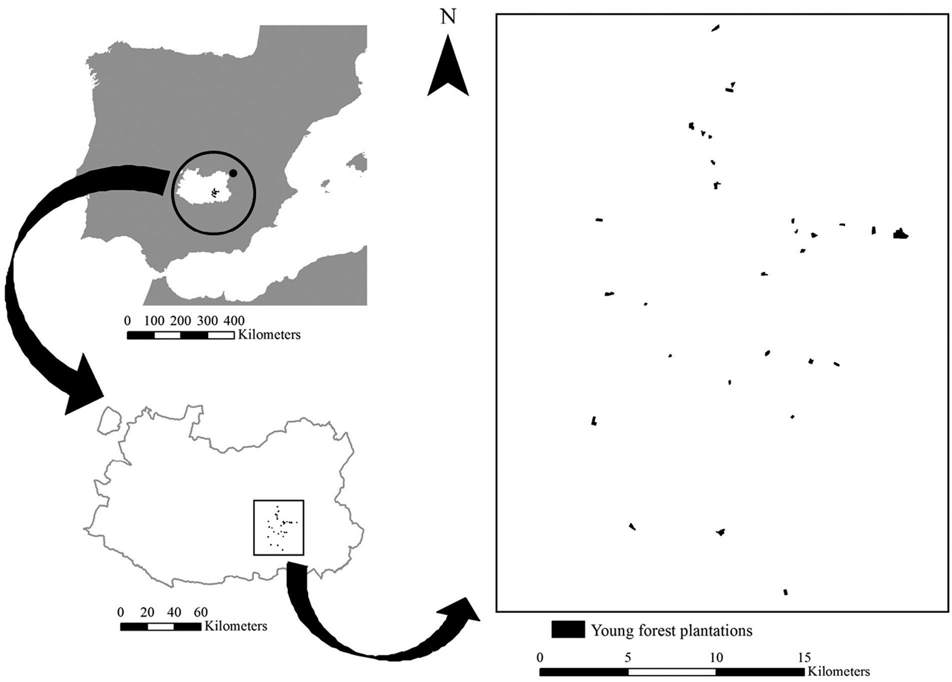

Figure 1. Location of the study area (rectangle) within the Ciudad Real province (white area) in central Spain and distribution of the 31 tree plantations that were investigated in this study. A circle of radius = 150 km is centered at the baricenter of the study area, and has been used to select the UTM squares of 10x10 km^2 that have been considered to quantify the regional distribution area of each species (number of occupied UTM squares). The black dot on the gray map of the Iberian Peninsula shows the location of the study area of Díaz *et al.* (1996).

The area is a representative mosaic of different crops and semi-natural or introduced woody vegetation patches that are characteristic of large extensions of Mediterranean landscapes. Croplands were mostly occupied by herbaceous crops (wheat and barley), harvested once a year in June, and permanent woody crops (olive trees — 3 to 5 m high, and vineyards — 1 m high). Natural vegetation typically consisted of dense Holm Oak *Quercus rotundifolia* L. woodland and riparian forests that have been mostly extirpated from this region. Until 1992, woodland cover was restricted to open Holm Oak woodlands, usually grazed by sheep and goats. However, as in many other Mediterranean landscapes, the agricultural land is subjected to intensive management (e.g., irrigation of vineyards and olive groves) and land use change. A major result of land use change is the abandonment of herbaceous cropland and vineyard extirpation and their afforestation with the native Aleppo pine *Pinus halepensis* alone or mixed with holm oak or *Retama sphaerocarpa*, which has increased forest land in the last 20 years. The relative extent of major land use types according to our ortophoto analysis (taken from SigPac Geographic Information System of Farming Land, http://www.magrama.gob.es/es/agricultura/temas/sistema-de-informacion-geografica-de-parcelas-agricolas-sigpac-/, in 1-km radius circles around the center of the 31 tree plantations studied here, 97.4 km^2 in total) were the following: olive grove (18.1% of the total land area), vineyard (22.3%), dry herbaceous cropland (19.7%), and scrubland (10.2%). We identified ten additional land use types, namely, waste lands, roads and rural tracks, vineyard with olive trees, woodland, urban areas and scattered buildings, fruit groves, pasture land, pastures with scattered trees, streams, rivers and lagoons, and dried-fruit orchards, each representing between 0.3 and 7.1% of the total area. Particularly, tree plantations and woodland spread on 3.1% and 0.8%, respectively, of the above mentioned 1-km radius circles (**Table 1**). A previous study found that 85% of the ortophoto identification coincided with field observations of checking points (Moreno-Mateos *et al.*, 2011).

Bird censuses

First, all pine plantations in the study area were located using both ortophotos (see source above) and Google Earth®, and were later verified in the field. We found 100 pine plantations that took place in 1992 or later. Next, we selected the plantations to be surveyed for birds, considering those with pines taller than 3.5 m and area larger than 1 ha (31 plantations; **Figure 1**), which were 12-18 years old.

Details on habitat structure of the 31 study pine plantations are shown in **Table 1**.

To asses bird occurrence in the novel habitat defined by the archipelago of pine plantations, bird censuses were carried out in spring (April and May) 2011. We did not intend to exhaustively census all the area covered by each single studied plantation because our goal was not to characterize species richness, but to establish a protocol to quantify species occurrence in the investigated novel habitat (for details on spatial variation of bird species richness with area of pine plantations in the study region, see Díaz, Asensio & Tellería, 1996, and Santos *et al.*, 2006). Thus, only one census plot was established in each one of the 31 pine plantations, in order to avoid pseudoreplication when quantifying relative abundance of bird species in this novel habitat. We assessed the occurrence of

Table 1 Mean, standard deviation (SD) and range (min / max) of the habitat structure variables in 31 tree plantations and landscape cover variables around such plantations in Campo de Montiel (La Mancha, Central Spain) during spring 2011.

	Mean	SD	Range	
Habitat structure				
Area (ha)	4.6	3.7	1.3	21.9
Tree layer cover (%)	50.6	24.1	15.3	100.0
Average tree height (m)	4.7	1.0	3.5	7.2
Density of trunks 10-20 cm dbh (# in 0.2 ha)	79.5	42.2	16.0	168.0
Density of trunks >20 cm dbh (# in 0.2 ha)	2.7	5.7	0.0	26.0
Cover of shrubs (%)	2.8	7.2	0.0	31.7
Average shrub height (m)	0.9	1.0	0.0	2.9
Cover of the herbaceous layer (%)	46.1	39.2	0.0	100.0
Landscape cover around plantations (%)				
Streams. rivers and lagoons	0.8	1.3	0.0	4.3
Roads and rural tracks	7.1	5.2	0.2	22.1
Woodlands	3.9	4.1	0.4	18.9
Holm oak Woodland	0.8	1.5	0.0	7.4
Pine plantations	3.1	2.6	0.4	11.5
Fruit groves	1.1	1.3	0.0	5.3
Waste lands	7.1	4.2	0.0	13.3
Olive groves	18.1	21.8	0.0	68.9
Pastures with scattered trees	0.3	0.4	0.0	2.0
Scrubland	10.2	7.3	0.0	28.4
Pastures	1.3	1.6	0.0	7.7
Dry herbaceous cropland	19.7	8.8	0.0	37.7
Vineyards	22.3	14.2	0.0	45.4
Vineyards with olive trees	5.4	9.6	0.0	33.6
Dried fruit orchards	0.8	2.3	0.0	9.5
Urban areas and scattered buildings	1.8	1.9	0.0	6.7

bird species in these census plots using point-count stations (Bibby *et al.*, 2000) lasting 10 minutes. All auditory and visual contacts were recorded, but only those within a 50 m radius (0.78 ha) were used in subsequent analyses, in order to increase the detection probability of studied species. Every census plot was surveyed twice on different days, once in the morning between sunrise and three hours later and once in the evening two hours before sunset. All censuses were conducted by the same well trained field ornithologist (JSSO) on windless (wind speed < 3 m s^{-1}) and rainless days. Nocturnal birds, aerial feeders such as swallows or swifts and raptors were not considered in data analyses, as this census method does not accurately estimate the occurrence of these species. A species was considered to be present in the census plot if it was detected in at least one of the censuses. The cumulative census time of 20 min in the two censuses carried out in each plot defines a long time devoted to bird census per unit of area, maximizes the detection probability of species and, thus, the accurate estimations of their occurrence in the habitat provided by the studied plantations (Shiu & Lee, 2003).

Regional patterns of bird distribution

Three data sources were used to characterize the patterns of distribution, habitat preferences and population trends of common birds in the region around the study area. First, the distribution area of each species 150 km around the geographical centre of the study area was obtained from the National Breeding Bird Atlas (Martí & del Moral, 2003) as the proportion of occupied 10 x 10 UTM km squares (308 UTM squares in total). The region defined by this circle includes the study area where Díaz *et al.* (1996)) carried out the analysis of bird occupation of mature pine plantations during the breeding season (ca. 85 km to northeast).

Secondly, habitat breadth of the bird species in 15 main habitat categories as well as their relative abundance in woodlands, agricultural areas and urban environments within the Mesomediterranean region of Central Spain were obtained from Carrascal & Palomino (2008; electronic Appendix: http://avesbiodiv.mncn.csic.es/19mono-suppl.pdf). Habitat breadth was calculated using Levins' (1968) index, divided by the number of habitat categories considered. This index ranges between 1 (evenly distributed across the 15 habitats) and 1/15 (only present in one habitat). Relative abundances for each species were calculated by dividing the measured densities provided by Carrascal & Palomino (2008) in each habitat by the maximum regional density recorded in the 15 main habitats of the Mesomediterranean region of Central Spain (considering the maximum density measured in three types of forests –pine, holm oak and deciduous woodlands–, five types of agricultural habitats –dry arable lands, irrigated lands, vineyards, olive groves and agricultural mosaics with woody cultivations–, and two types of urban habitats –towns and periurban developments with scattered buildings); relative abundances range between 1 (maximum density attained at that habitat) and 0 (absent). Preference of bird species for mature pine plantations in the study region was obtained from Díaz *et al.*, (1996), and was estimated as the proportion of occupied plantations (n=48).

And third, the Spanish SACRE programme (monitoring of common breeding birds in Spain) was used to quantify the population changes of the studied species from 1998 to 2011 in Central Spain (SEO/Birdlife, 2012). Population changes were measured as the percentage of change in 2011 respect to the 1998 baseline data (see **Online Resource 1**).

Morphological and life-history traits of birds

Data on body mass was obtained from Cramp (1998), and information on clutch size from Lislevand, Figuerola & Székely (2007). The migratory strategies of birds (trans-Saharan migrant, score 0 vs. resident species not migrating outside the Iberian Peninsula, score 1) in the study area were taken from Díaz *et al.* (1996) and Tellería *et al.* (1999); see **Online Resource 1** for details.

Data analyses

The number of occupied afforested plots by each bird species was used as a measure of their occurrence in the novel habitat defined by the archipelago of pine plantations. The interspecific variation in occurrence was analyzed using a generalized linear model with negative binomial errors and the log-link function (Crawley, 1993). In this kind of model, the response variable represents a count of the occurrence and must have only non-negative integer values, and the conditional variance of the count is given by $\mu(1+\alpha)$, where μ denotes the conditional mean and α is calculated by the model using the maximum-likelihood (ML) estimation. Statistical significance of the predictor variables (see below) was estimated using a robust approach where quasi-ML standard errors are calculated using a "sandwich" of the inverse of the Hessian and the Outer Product of the Gradient (Lindsey, 2004; Cottrell & Lucchetti, 2011). The negative binomial generalized linear model is a good solution for zero-inflated Poisson models where the over-dispersion parameter ϕ is highly deviated from one. In fact, the Poisson regression model produced a higher AIC figure (231.5) and a poorer residual plot than the negative binomial model (AIC = 183.4). Standardized regression coefficients (β) were obtained in the regression analysis (i.e., analysis was performed with standardized variables, so that their averages were zero and variances were 1). Statistical analyses were carried out using Gretl package 1.9.5cvs (http://gretl.sourceforge.net/).

In order to compare the explanatory power of different subsets of variables, four alternative *a priori* models were compared by means of Akaike's second-order AIC corrected for small sample sizes (AICc), and their derived weights (Burnham & Anderson, 2002): (1) random sampling effects (proportion of occupied 10 x 10 UTM km squares around the study area, habitat breadth and population trend of the species), (2) regional habitat preferences (for mature pine plantations, woodland, agricultural and urban environments), (3) life history traits (migratory strategy, body mass and clutch size) and (4) combined effects of variables included in models (2) and (3).

It is commonly acknowledged that species are evolutionarily related throughout a phylogenetic scheme, and therefore they should not be treated as independent sample units in comparative analyses (Harvey & Purvis, 1991; but see Westoby, Leishman & Lord, 1995; Price, 1997). Nevertheless, we are not interested in patterns of biological diversification throughout evolutionary time in this particular study, but only in present-day relationships pertaining to the occurrence of species in an intensively human-transformed environment. Thus, we simplified the data analyses by avoiding the complexity and drawbacks of comparative methods (i.e., uncertainty about models of evolutionary change, phylogeny topology or branch lengths).

Results

Twenty-four out of 80 terrestrial bird species of the study region were observed in the 31 tree plantations that were studied. The most widespread species were the wood pigeon *Columba palumbus* and the goldfinch *Carduelis carduelis*, which occurred in more than 80% of the pine plantations. The magpie *Pica pica* and the great tit *Parus major* were also relatively frequent, occurring in more than one-third of the pine plantations. The remaining species were relatively scarce, and 12 species were present in less than one-tenth of the plantations (**Online Resource 1**). Many species with marked habitat preferences for woodland habitats were very scarce (except the great tit) or were never recorded in the pine plantations (e.g. firecrest *Regulus ignicapilla*, short-toed treecreeper *Certhia brachydactyla*, long-tailed tit *Aegithalos caudatus*, nuthatch *Sitta europaea*, crested tit *Lophophanes cristatus*, great spotted woodpecker *Dendrocopos major* and European jay *Garrulus glandarius*).

Occurrence of species in pine plantations was significantly explained by a model including the ten predictor variables (χ^2_{10} = 62.34, P < 0.001; Nagelkerke pseudo R^2 = 0.640; AICc = 183.4), and was significantly and positively associated with the proportion of occupied 10 x 10 UTM km squares around the study area, habitat breadth, the population trend of species within the period 1998-2011, and the occupation index of mature pine plantations in the study region (**Table 2 and Figure 2**). The extent of regional distribution and habitat breadth were the two predictor variables with the highest magnitude effects according to the standardized regression coefficients. The remaining predictor variables were not significantly related to the occurrence of species in the pine plantations (P > 0.24; **Table 2**).

Table 2 Generalized linear regression model (with negative binomial errors and the log-link function) relating the habitat occupancy of 80 bird species in young and small pine plantations of agricultural areas of Central Spain and ten predictor variables describing regional distribution, habitat breadth, population trends (1998-2011), regional habitat preferences (four variables), migratory strategy, body mass, and clutch size. p: Statistical significances were estimated using a robust approach with quasi-ML standard errors. Regional distribution: proportion of occupied 10 x 10 UTM km squares 150 km around the study area. Occupation index of mature plantations: proportion of occupied plantations in southern xeric Iberian plateau obtained from Díaz *et al.* (1996). Migratory strategy: 1 - resident. 0 - trans-Saharan migrant. Coefficient: standardized regression coefficients that inform about the magnitude and sign of the partial relationships of the predictor variables and the response variable. The meaning of the rest of the variables is described in the Methods section; data for the 80 studied species can be found in **Online Resource 1**.

	Coefficient	SE	p
Regional distribution	1.03	0.53	0.050
Habitat breadth	0.94	0.36	**0.009**
Population trend (1998-2011)	0.39	0.12	**<0.001**
Occupation index of mature plantations	0.25	0.10	**0.009**
Relative abundance in woodlands	-0.08	0.31	0.806
Relative abundance in agricultural habitats	-0.07	0.23	0.751
Relative abundance in urban environments	-0.03	0.22	0.903
Migratory strategy	0.29	0.24	0.236
Body mass (in ln)	-0.02	0.16	0.879
Clutch size	0.06	0.18	0.714

The model including only the predictor variables related to the extent of regional distribution, habitat breadth and population trends of species had the lowest Akaike Information Criterion (AICc = 175.4; Akaike weight ≈ 1), and thus the highest strength of evidence, in comparison to the models including life history traits (migratory strategy, body mass and clutch size; AICc = 226.3), habitat preferences (AICc = 201.6) or the combination of these two subsets of predictor variables (AICc = 291.4; weights ≈ 0 for these three models).

Discussion

This study clearly shows that woodland restoration based on small, highly fragmented, pine plantations in a Mediterranean landscape matrix dominated by agricultural habitats does not contribute to enhancing avian diversity by capturing woodland birds, especially if the natural forests of the region do not belong to the coniferous vegetation domain. Under these circumstances, the chance of encountering a particular species in pine plantations is the consequence of passive, random sampling, instead of ecological processes mediated by habitat preferences or life history traits, where the most widespread species with broader habitat preferences (eurytopic taxa) and increasing population trends are favored.

The low influence of autoecological traits of species in determining the probability of occurrence in pine plantations is reinforced by two additional characteristics of the studied region and plantations: the low maturity and small size of pine plantations, and the low favorability of the region for the forest avifauna considering biogeographic constraints (Tellería & Santos, 1993; Carrascal & Díaz, 2003). The young and small isolated pine plantations, within a matrix of deforested agricultural landscape (<1% of broadleaf dry sclerophyllous forests in our study area), reduce their attractiveness for woodland specialist species, considering the fragmentation of populations as several studies have shown (e.g. Díaz *et al.*, 1998; Izhaki, 1999; Santos, Tellería & Carbonell, 2002; Santos *et al.*, 2006, for the Mediterranean region). Moreover, and as Shochat *et al.* (2001) have shown with pine plantations in Israel, pine plantations in Mediterranean zones are generally too simplistic in structure to maintain rich bird communities, mainly due to the lack of suitable microhabitats in the understory (see also Lopez & Moro, 1997). On the other hand, the studied plantations are located in a region with impoverished forest avifauna dominated by species of Mediterranean origin with marked preferences for sclerophyllous shrublands or open woodlands

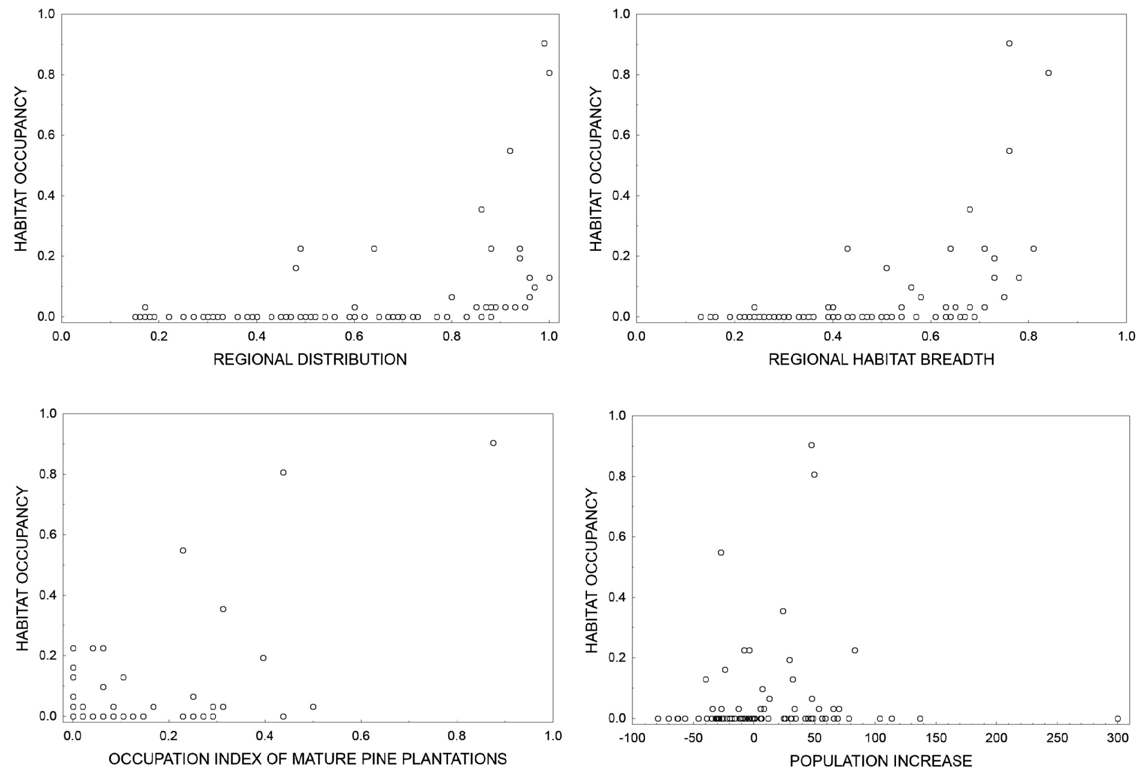

Figure 2. Relationship between interspecific variation in habitat occupancy of young and small pine plantations of agricultural areas of Central Spain and regional distribution (proportion of occupied 10x10 UTM km squares 150 km around the study area), population trends (percentage of change from 1998 to 2011), regional habitat breadth, and occupation index of mature plantations (proportion of occupied plantations in southern xeric Iberian plateau obtained from Díaz *et al.* (1996). Habitat occupancy is the number of occupied census plots where the species were present, divided by the total number of censused plots. Sample size is 80 bird species.

(Monkkonen, 1994; Tellería & Santos, 1994; Carrascal & Díaz, 2003). Coniferous forest specialists, such as the firecrest *Regulus ignicapillus*, the crested tit *Lophophanes cristatus*, the coal tit *Periparus ater*, the nuthatch *Sitta europaea* or the crossbill *Loxia curvirostra*, which require larger and more mature woodland patches (Díaz *et al.*, 1998; Santos *et al.*, 2006), were very scarce or never recorded in the region, thus emphasizing the low favourability of the study area for forest avifauna of the coniferous domain. This observed pattern reinforces the importance of the biogeographic context when designing restoration plans based on afforestations in agricultural landscapes (Suárez-Seoane, Osborne & Baudry, 2002).

The pattern of bird species occurrence in the mosaic of pine plantations surrounded by cropland arose just as a consequence of probabilistic reasons related to the abundance and population trends of the species at a regional scale: those species occupying greater proportions of territory around the study area, exhibiting larger habitat breadth and with increasing population trends were those most frequently encountered in the plantation plots. Thus, the commonness of bird species in the study region determine their occupation of pine plantations, a result that agrees with the rather common positive relationship between regional abundance and distribution of species in many animal groups (Gaston, 1994). The tight relationship between regional habitat breadth and occupancy of this novel, highly fragmented habitat, is consistent with the value of niche-based characteristics of species in explaining patterns of bird distribution from the level of local habitats to that of geographical ranges (see also, (Swihart *et al.*, 2003; Bohning-Gaese *et al.*, 2006; Hurlbert & White, 2007; Carrascal *et al.*, 2008; Slatyer, Hirst & Sexton, 2013).

By contrast, bird species that were expected to occur most frequently in the pine plantations due to their preferences for woodland habitats and avoidance of agricultural and urban habitats (Santos *et al.*, 2006; Sirami *et al.*, 2008a, 2008b; Rey Benayas *et al.*, 2010) did not exhibit a higher frequency than species with other habitat preferences. Only the occupancy of mature pine afforestations in the same region had a relevant influence on the frequency of occurrence of birds in the small and highly fragmented plantations of agricultural areas of Central Spain, although its magnitude effect was relatively low according to its standardized regression coefficient (see **Table 2**). Similarly, species that were expected to be good colonizers of novel habitats because of characteristics like sedentariness, large clutch size and small body size (Galván & Rey Benayas, 2011), did not occur more frequently in the studied plantations than species with other autoecological traits. Therefore, our analyses indicate that the occurrence of birds in pine plantations in abandoned Mediterranean cropland is explained by the effects of random sampling, but it is poorly related to the habitat preferences and autoecological traits of the bird species. This pattern is probably the consequence of the small area of the pine plantations in the study region, determined by the scheme of land tenure of small agricultural properties. In fact, the small area of plantations, with an average of 4.6 ha and ranging between 1 and 22 ha, is considerably lower than the minimum area requirements of many woodland specialists in Central Spain that need more than 10 ha to be present, such as the stock dove *Columba oenas*, the great spotted woodpecker *Dendrocopos major*, the orphean warbler *Sylvia hortensis*, the golden oriol *Oriolus oriolus*, the Eurasian jay *Garrulus glandarius* or the cirl bunting *Emberiza cirlus* (Díaz *et al.*, 1998).

However, it would not be correct to design restoration strategies based on pine plantations ignoring the autoecological traits of the bird species just because they cannot predict bird occurrence. Some autoecological traits of birds, in particular migratory strategy, egg mass and body mass, have been shown to predict the density, not occurrence, of bird species in other pine plantations on an agricultural matrix located in a nearby study region (Galván & Rey Benayas, 2011). Bird species density in pine plantations were determined by ecological processes as expected by the fact that species that are sedentary and have small egg and body masses are good colonizers of novel habitats (Cassey, 2001; Duncan, Blackburn & Sol, 2003; Galván & Rey Benayas, 2011). It may thus be possible that two different indicators of the success of pine plantations in gathering bird populations, namely species occurrence and local density, are determined by different processes, the former being dependent on the effects of random sampling and the latter responding to the autoecological traits of species. This suggestion, however, must be taken with caution because the afforested fields of the present study are considerably smaller (mean area = 4.6 ha) than those in which bird density was found to be associated with autoecological traits of the species (area > 25 ha; Galván & Rey Benayas, 2011), so processes controlling their colonization by birds cannot be straightforward compared. Future studies should investigate if bird occurrence and density are actually dependent on different factors in Mediterranean cropland afforestations.

In conclusion, pine plantations favored by the European CAP resulting in an archipelago of man-made woodland islands within a Mediterranean agricultural landscape, only capture widespread and habitat generalist avian species with increasing population trends, not contributing to favouring truly woodland species. This result casts doubts on the value of this restoration practice for the conservation and management of avian diversity in the Mediterranean region if it is developed in very small woodland areas considering the pattern of land tenure of small properties.

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Online Resource 1

Species data for the terrestrial avifauna of the study area (Campo de Montiel, La Mancha, Central Spain). OCCURRENCE: number of occupied pine plantation plots out of 31. REG-DIST: proportion of occupied 10 x 10 UTM km squares 150 km around the geographical centre of the study area (obtained from Martí & del Moral, 2003). AH15: habitat breadth of the bird species in 15 main habitat categories the Mesomediterranean region of Central Spain (obtained from Carrascal & Palomino, 2008). POP-INCR: population change in Central Spain measured as the percentage of change in 2011 respect to 1998 baseline data (obtained from SEO/Birdlife, 2012). OCC-PLANT: proportion of occupied plantations in southern xeric Iberian plateaux obtained from Díaz *et al.* (1996). WOODLAND, AGRICULT and URBAN: habitat preference for woodlands, agricultural habitats and urban environments of Central Spain, respectively (1: maximum; 0: not recorded in those habitats; obtained from Carrascal & Palomino, 2008). MIGRAT: migratory strategy of birds (trans-Saharan migrant, score 0 vs. resident species not migrating outside the Iberian Peninsula, score 1; obtained from Díaz *et al.*, 1996, and Tellería *et al.*, 1999). MASS: bird mass in g (obtained from Cramp, 1998). CLUTCH: clutch size (obtained from Lislevand *et al.*, 2007).

SPP	OCURRENCE	REG-DIST	AH15	POP-INCR	OCC-PLANT	WOODLAND	AGRICULT	URBAN	MIGRAT	MASS	CLUTCH
Acrocephalus arundinaceus	0	0.31	0.22	-3.2	0.00	0.03	0.97	0.13	0	31.0	4.5
Acrocephalus scirpaceus	0	0.39	0.19	113.4	0.00	0.02	0.38	0.04	0	12.5	4.0
Aegithalos caudatus	0	0.60	0.27	-35.0	0.23	1.00	0.09	0.12	1	7.5	10.0
Alauda arvensis	5	0.48	0.51	-24.0	0.00	0.13	0.69	0.06	1	38.0	4.0
Alectoris rufa	4	1.00	0.78	-39.8	0.10	0.37	0.90	0.32	1	477.5	13.0
Anthus campestris	0	0.36	0.21	-6.2	0.00	0.02	0.11	0.22	0	23.5	4.5
Burhinus oedicnemus	0	0.72	0.29	-29.4	0.00	0.04	0.43	0.06	1	410.0	1.9
Calandrella brachydactyla	0	0.56	0.40	-1.5	0.00	0.05	0.67	0.22	0	22.3	4.0
Carduelis cannabina	6	0.94	0.73	29.2	0.40	0.31	0.70	0.30	1	17.6	4.6
Carduelis carduelis	25	1.00	0.84	49.8	0.44	0.60	1.00	0.72	1	16.0	4.2
Carduelis chloris	1	0.95	0.68	69.7	0.17	0.35	1.00	0.91	1	26.5	4.3
Certhia brachydactyla	0	0.59	0.46	25.7	0.00	1.00	0.36	0.49	1	8.3	5.2
Cettia cetti	0	0.54	0.44	-10.7	0.00	0.18	0.42	0.30	1	13.3	4.5
Cisticola juncidis	0	0.46	0.34	-28.7	0.00	0.06	0.46	0.10	1	8.9	3.3
Clamator glandarius	7	0.64	0.43	83.4	0.04	0.11	0.20	0.23	0	155.0	4.8
Coccothraustes coccothraustes	0	0.18	0.16	-11.8	0.04	1.00	0.28	0.00	1	56.5	4.5
Columba livia	1	0.88	0.63	5.4	0.00	0.09	0.60	1.00	1	270.0	2.0
Columba oenas	0	0.16	0.43	-4.4	0.04	0.20	0.59	1.00	1	295.0	2.3

SPP	OCURRENCE	REG-DIST	AH15	POP-INCR	OCC-PLANT	WOODLAND	AGRICULT	URBAN	MIGRAT	MASS	CLUTCH
Columba palumbus	28	0.99	0.76	47.2	0.88	0.45	0.45	0.90	1	507.0	1.9
Corvus corax	0	0.40	0.52	0.6	0.00	0.29	0.45	0.18	1	1250.0	5.0
Corvus corone	0	0.32	0.41	-26.8	0.23	0.36	0.24	0.05	1	570.0	4.5
Corvus monedula	0	0.59	0.47	-57.1	0.00	0.17	0.40	0.27	1	240.0	5.0
Coturnix coturnix	1	0.87	0.40	-27.1	0.00	0.20	0.51	0.00	0	101.4	10.5
Cuculus canorus	0	0.67	0.54	30.1	0.08	0.69	0.25	0.09	0	110.8	9.2
Cyanistes caeruleus	2	0.80	0.58	47.6	0.00	1.00	0.32	0.08	1	11.3	7.0
Cyanopica cooki	0	0.51	0.29	65.5	0.00	0.67	0.13	0.02	1	72.0	6.0
Dendrocopos major	0	0.40	0.39	34.4	0.08	1.00	0.16	0.27	1	74.4	5.5
Emberiza calandra	4	0.96	0.73	32.0	0.00	0.67	0.83	0.21	1	43.0	4.9
Emberiza cia	0	0.45	0.39	-31.4	0.00	1.00	0.20	0.12	1	23.5	4.1
Emberiza cirlus	0	0.38	0.33	44.5	0.02	0.17	0.13	0.00	1	25.5	3.5
Erithacus rubecula	0	0.30	0.29	47.3	0.00	0.36	0.09	0.06	1	16.7	5.2
Fringilla coelebs	1	0.85	0.54	53.5	0.50	0.76	1.00	0.17	1	23.0	4.0
Galerida cristata	7	0.88	0.71	-7.9	0.00	0.36	0.99	0.20	1	41.4	4.1
Galerida theklae	0	0.62	0.28	30.8	0.00	0.14	0.07	0.04	1	36.8	4.1
Garrulus glandarius	0	0.45	0.48	-7.6	0.06	1.00	0.30	0.12	1	174.0	5.7
Hippolais polyglotta	0	0.60	0.67	24.6	0.00	0.20	0.34	0.25	0	11.0	4.5
Jynx torquilla	0	0.15	0.34	5.6	0.00	0.13	0.69	0.36	0	37.5	8.7
Lanius meridionalis	0	0.70	0.63	-63.1	0.00	0.59	0.83	0.17	1	63.5	5.5
Lanius senator	1	0.89	0.54	65.1	0.08	1.00	0.57	0.05	0	30.5	4.9
Lophophanes cristatus	0	0.29	0.21	-17.9	0.00	1.00	0.05	0.06	1	10.5	6.5
Lullula arborea	0	0.68	0.39	11.9	0.25	0.46	0.19	0.06	1	26.1	4.0
Luscinia megarhynchos	0	0.77	0.61	42.2	0.08	0.23	0.39	0.29	0	20.2	4.8
Melanocorypha calandra	0	0.65	0.27	-22.3	0.00	0.05	0.46	0.03	1	65.0	4.2
Monticola solitarius	0	0.33	0.36	6.0	0.00	1.00	0.24	0.48	1	56.5	4.5
Motacilla alba	0	0.47	0.64	-19.8	0.00	0.35	0.42	0.56	1	21.0	5.5
Motacilla cinerea	0	0.27	0.27	-70.3	0.00	0.11	0.32	1.00	1	18.0	5.0
Motacilla flava	0	0.19	0.19	48.5	0.00	0.00	1.00	0.27	0	16.3	5.4
Muscicapa striata	0	0.31	0.35	-79.2	0.00	0.78	0.25	1.00	0	15.0	4.4
Oenanthe hispanica	0	0.88	0.24	-30.2	0.00	0.05	0.14	0.08	0	17.9	5.3
Oenanthe leucura	0	0.22	0.25	-30.5	0.00	0.43	0.07	0.11	1	38.0	4.0

SPP	OCURRENCE	REG-DIST	AH15	POP-INCR	OCC-PLANT	WOODLAND	AGRICULT	URBAN	MIGRAT	MASS	CLUTCH
Oenanthe oenanthe	0	0.33	0.28	-25.3	0.00	0.04	0.16	0.15	0	24.4	5.5
Oriolus oriolus	0	0.79	0.51	56.4	0.13	0.37	0.20	0.09	0	70.0	3.5
Parus major	11	0.86	0.68	23.9	0.31	1.00	0.65	0.24	1	16.8	7.5
Passer domesticus	3	0.97	0.56	6.8	0.06	0.14	0.27	1.00	1	28.0	4.0
Passer hispanicus	0	0.19	0.31	55.9	0.00	1.00	0.53	0.03	1	27.0	5.0
Passer montanus	1	0.60	0.39	-34.5	0.02	0.06	0.59	0.35	1	22.0	4.5
Periparus ater	1	0.17	0.24	-12.6	0.00	1.00	0.06	0.16	1	9.9	5.7
Petronia petronia	0	0.62	0.57	58.8	0.13	0.46	0.62	0.27	1	31.0	5.5
Phoenicurus ochruros	0	0.43	0.50	-0.4	0.00	0.34	0.18	0.35	1	16.5	4.9
Phylloscopus bonelli	0	0.25	0.23	113.4	0.00	0.60	0.05	0.10	0	8.1	4.4
Pica pica	17	0.92	0.76	-27.5	0.23	0.35	0.62	0.50	1	225.0	6.1
Picus viridis	0	0.86	0.66	-62.7	0.29	0.47	0.27	0.54	1	175.5	6.0
Regulus ignicapilla	0	0.17	0.15	42.4	0.00	0.19	0.03	0.05	1	5.3	8.0
Saxicola torquata	0	0.69	0.69	6.1	0.00	0.31	0.52	0.28	1	15.2	5.0
Serinus serinus	1	0.93	0.65	8.3	0.31	0.84	1.00	0.53	1	11.5	3.7
Sitta europaea	0	0.22	0.22	103.7	0.00	1.00	0.14	0.00	1	23.4	6.5
Streptopelia decaocto	7	0.49	0.64	828.2	0.00	0.76	0.44	1.00	1	188.0	1.9
Streptopelia turtur	0	0.83	0.54	-39.0	0.44	0.61	1.00	0.16	0	144.0	1.9
Sturnus unicolor	7	0.94	0.81	-3.7	0.06	0.46	0.67	1.00	1	88.0	4.0
Sylvia atricapilla	0	0.52	0.48	78.6	0.00	0.34	0.34	0.40	1	22.3	4.5
Sylvia cantillans	0	0.50	0.36	68.8	0.27	0.48	0.07	0.39	0	9.5	4.2
Sylvia communis	0	0.18	0.13	-15.6	0.00	0.16	0.03	0.00	0	14.8	4.5
Sylvia conspicillata	0	0.32	0.22	136.7	0.00	0.10	0.05	0.08	0	9.5	4.0
Sylvia hortensis	0	0.30	0.33	299.9	0.13	0.28	0.31	0.06	0	22.0	4.2
Sylvia melanocephala	0	0.73	0.43	-9.5	0.00	0.67	1.00	0.21	1	11.2	4.2
Sylvia undata	0	0.62	0.61	-45.9	0.10	0.77	0.26	0.49	1	10.5	4.0
Troglodytes troglodytes	0	0.49	0.30	-6.5	0.00	0.34	0.07	0.05	1	8.8	6.5
Turdus merula	1	0.91	0.71	33.4	0.29	0.63	0.52	0.46	1	86.1	3.5
Turdus viscivorus	0	0.60	0.26	-4.1	0.15	0.24	0.08	0.12	1	119.2	3.5
Upupa epops	2	0.96	0.75	12.8	0.25	1.00	0.67	0.17	0	68.6	4.4

Foto: Ganga ibérica (*Pterocles alchata*)

Autor: Juan S. Sánchez Oliver

Capítulo 4

Repulsión de las aves de zonas agrícolas por las plantaciones forestales jóvenes en campos de cultivo mediterráneos

Este capítulo reproduce íntegramente el texto del siguiente manuscrito:

Sánchez-Oliver, J.S., Rey Benayas, J.M. & Carrascal, L.M. (2013). Repellence of open-farmland birds by young afforestation of Mediterranean cropland. En revisión.

Resumen

Los programas de reforestación como el promovido por la Política Agraria Comunitaria de la UE han extendido las plantaciones forestales en campos anteriormente cultivados. Estas reforestaciones pueden causar graves daños a las especies de hábitats abiertos, especialmente a aves con alto valor de conservación. Investigamos los efectos que las plantaciones forestales jóvenes (<20 años y dominadas por pinos) del centro de España tienen sobre las comunidades de aves que ocupan los hábitats agrícolas abiertos adyacentes. Planteamos que las zonas más próximas a las plantaciones de pinos con mayor superficie tendrán menores valores de riqueza de especies y valor de conservación a nivel Europeo, y en ellas será menor la abundancia relativa de especies de aves propias de áreas agrícolas. Controlando la influencia de las categorías de usos del suelo alrededor de las plantaciones, se obtuvieron efectos significativos de la distancia desde el borde de las plantaciones de pinos y del área de la plantación sobre la riqueza de especies de aves en invierno pero no en el periodo reproductor, y en la composición de la comunidad en el periodo reproductor pero no en el invierno. La distancia desde el borde de las plantaciones forestales mostró efectos significativos en un índice del estado de conservación de las especies (SPEC) en el periodo reproductor pero no en invierno. En invierno, *Columba livia*, *Pica pica*, *Pterocles orientalis* y *Serinus serinus* mostraron mayores frecuencias lejos de las plantaciones forestales; sin embargo, *Parus major* –una especie forestal generalista– fue más frecuente a distancias cercanas a las plantaciones. En el periodo reproductor, el área de las plantaciones forestales tuvo un efecto detrimental en la frecuencia de *Alectoris rufa*, *Streptopelia decaocto* y *Sturnus unicolor*, pero *Tetrax tetrax* alcanzó una frecuencia mayor en los transectos localizados cerca de las plantaciones más grandes. Las plantaciones más grandes tuvieron un efecto detrimental en la frecuencia de *A. rufa.* Concluimos que la reforestación con pinos de campos de cultivo mediterráneos en paisajes agrícolas heterogéneos tiene un efecto perjudicial leve, particularmente en invierno, en las especies de aves que son características de hábitats agrícolas abiertos y apenas favorece a las especies que son forestales generalistas o ubiquistas.

Palabras clave Composición de la comunidad - Estado de conservación – Efectos de la distancia – Tipos de usos del suelo – Plantaciones de pinos – Presencia de especies – Riqueza de especies

Repellence of open-farmland birds by young afforestation of Mediterranean cropland

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Abstract

Afforestation programs such as the one promoted by the EU Common Agricultural Policy have spread tree plantations on former cropland. These afforestations may cause severe damage to open habitat species, especially birds of high conservation value. We investigated the effects of young (<20 yr) tree plantations dominated by pines on bird communities inhabiting the adjacent open farmland habitat in central Spain. We hypothesised that pine plantations with larger surface, and areas at shorter distances from plantations, would result in lower bird species richness, species frequency and conservation value of open farmland birds. Statistical models controlling for the influence of plantation area and land use categories around plantations revealed significant effects of distance from pine plantation edge on community species richness in winter (i.e., communities at >600 m from the plantations were ca. 30% richer in species that those at <400 m) but not in the breeding season, and on community composition in the breeding season but not in winter. Distance from tree plantation edge showed significant effects on an index of conservation concern (SPEC) during the breeding season (i.e., 19% higher conservation value close to tree plantations), but not in winter. Proportion of dry herbaceous cropland was the variable with the strongest effect in overall community composition. In winter, *Columba livia, Pica pica, Pterocles orientalis*, and *Serinus serinus* attained a higher frequency away from tree plantations; however, *Parus major* –a forest generalist species– was more frequent at shorter distances from tree plantations. In breeding season, tree plantation area had a detrimental effect on the frequency of *Alectoris rufa*, *Streptopelia decaocto* and *Sturnus unicolor*, but *Tetrax tetrax* attained a higher frequency in transects located at large plantations. Larger tree plantations had a detrimental effect on the frequency of *A. rufa*. We conclude that pine afforestation of Mediterranean cropland in heterogeneous agricultural landscapes has an overall low detrimental effect on bird species that are characteristic of open farmland habitat, and that hardly favour species that are forest generalist or ubiquitous species.

Keywords Community composition – Conservation status – Distance effects – Land use types – Pine plantations – Species occurrence – Species richness

Introduction

A significant amount of abandoned cropland, low productive cropland and pastureland has been converted into tree plantations in the last few decades, and ca. 7% of forest land in the world are tree plantations at present (FAO, 2011). Different afforestation programs have contributed to the spread of such tree plantations at the regional level. For instance, the Common Agrarian Policy (CAP) has favoured the conversion of farmland into tree plantations in the European Union since 1992 by means of a scheme of aid for forestry measures in agriculture (EEC Council Regulation No. 2080/92), which has resulted in the afforestation of ca. 8 millions ha to date (European Commission, 2013a, 2013b). Further, afforested cropland is expected to increase in the near future in countries such as Spain due to subsidies to afforestation of extirpated vineyards (Spanish Agrarian Guarantee Fund, 2012).

Tree plantations pursue a number of environmental and societal services such as soil retention and carbon sequestration (Rey Benayas *et al.*, 2007). However, they may have noticeable effects on biological communities. Thus, Bremer & Farley (2010) found that tree plantations are most likely to contribute to biodiversity when established on degraded lands rather than replacing natural ecosystems, and when indigenous tree species are used rather than exotic species. Similarly, a meta-analysis of faunal and floral species richness and abundance in timber plantations and pasture lands on 36 sites across the world concluded that plantations support higher species richness or abundance than pasture land only for particular taxonomic groups (i.e. herpetofauna), or specific landscape features (i.e. absence of remnant vegetation within pasture) (Felton *et al.*, 2010).

Agro-ecosystems are important for maintenance of bird diversity in Europe, especially for species of conservation concern (BirdLife International, 2004a). The Directorate-General for Agriculture and Rural Development (2012), using the European farmland bird index as a barometer of change for the biodiversity of agricultural land in Europe, shows a decline in these bird populations of ca. 20% between 1990 and 2008 (see also Donald *et al.*, 2002; Gregory *et al.*, 2005; Butler *et al.*, 2010; Scholefield *et al.*, 2011; Guerrero *et al.*, 2012). Cropland afforestations in southern Europe, which are mostly based on coniferous species, may cause severe damage to open habitat species, especially ground-nesting birds, by replacing high quality habitat and increasing risk of predation (Shochat, Abramsky & Pinshow, 2001; Santos *et al.*, 2006; Caplat & Fonderflick, 2009; Reino *et al.*, 2009, 2010; Butler *et al.*, 2010; Fonderflick *et al.*, 2010; Voříšek *et al.*, 2010). Many of such bird species are of conservation concern in Europe (European Bird Census Council, 2010).

Afforestation can increase predation risk through both direct and indirect effects on bird populations (Batáry & Báldi, 2004), the latter related to the avoidance of habitats that are perceived as risky (Murcia, 1995). Tree plantations act as sources of generalist predators of various types (Andren, 1992; Pita *et al.*, 2009; Reino *et al.*, 2010), which usually have very low densities at treeless open habitats, but thrive in mosaic habitat landscapes with tree plantations (Andren, 1992; Pita *et al.*, 2009; Reino *et al.*, 2010). Particularly, predation by corvids is enhanced in humanized landscapes where they attain high densities (Jokimaki, Huhta & Jokimäki, 2000; Newson *et al.*, 2010). Previous research on predation of bird nests at open habitat adjacent to tree plantations or forest fragments in Mediterranean agricultural landscapes has shown figures between 13.4 and 50% of predation rates (Santos & Tellería, 1992; Castilla *et al.*, 2007; Reino *et al.*, 2010). An assessment of nest predation rates on open farmland habitat adjacent to tree plantations in central Spain resulted in 94.2% of artificial nests that were predated three weeks after the start of the experiment (Sánchez-Oliver, Rey Benayas & Carrascal, 2013a).

In this study we aim at investigating the effects of young (<20 yr) tree plantations on bird communities inhabiting the adjacent open farmland habitat in a Mediterranean landscape mosaic located in central Spain. We hypothesize that (1) tree plantations define a surrounding (buffer) area that will have a detrimental effect on bird species that are characteristic of open farmland habitat, particularly for ground-nesting species that are of high conservation value. Thus, we predict that tree plantations with larger extension and those cropland areas located at shorter distances from tree plantations will result in lower frequency of open farmland birds, which will be most noticeable in the breeding season than in winter due to territorial behaviour. Similarly, we also hypothesize that (2) tree plantations may favour the occurrence and frequency of bird species that are forest generalist or ubiquitous species in the surrounding farmland landscape.

Methods

Study area

Field work was carried out on open farmland adjacent to afforested cropland located in Campo de Montiel (La Mancha natural region, southern Spanish plateau, UTM 30 S 469411 4289409; **Figure 1**). The study area spreads on ca. 440 km2 with altitude ranging between 690 and 793 m a.s.l. The climate is continental Mediterranean with dry and hot summers and cold winters. Mean annual temperature and total annual precipitation in the area during the last 30 years were 13.7 °C and 390 mm, respectively (Agencia Española de Meteorología, 2012). These figures were 15.8ºC and 362.9 mm in 2012, when our bird surveys took place (Junta de Castilla-La Mancha, 2013).

The area is a representative mosaic of different crops, pastures and semi-natural or introduced woody vegetation that are characteristic of large areas in Mediterranean landscapes. Croplands were mostly occupied by herbaceous crops (wheat and barley) and permanent woody crops (olive groves and vineyards). Natural vegetation consisted of holm oak (*Quercus rotundifolia* L.) woodland and riparian forests that have been mostly extirpated from this region. Until 1992, woodland cover was restricted to open holm oak patches, usually grazed by sheep and goats. Major land use changes in the last 20 years are the abandonment of herbaceous cropland and vineyard extirpation and their subsequent afforestation with the native Aleppo pine (*Pinus halepensis* Mill.) alone or mixed with holm oak. These tree plantations are of small area due to property size and noticeably dominated by pines as they establish better and grow faster than other planted species such as holm oak.

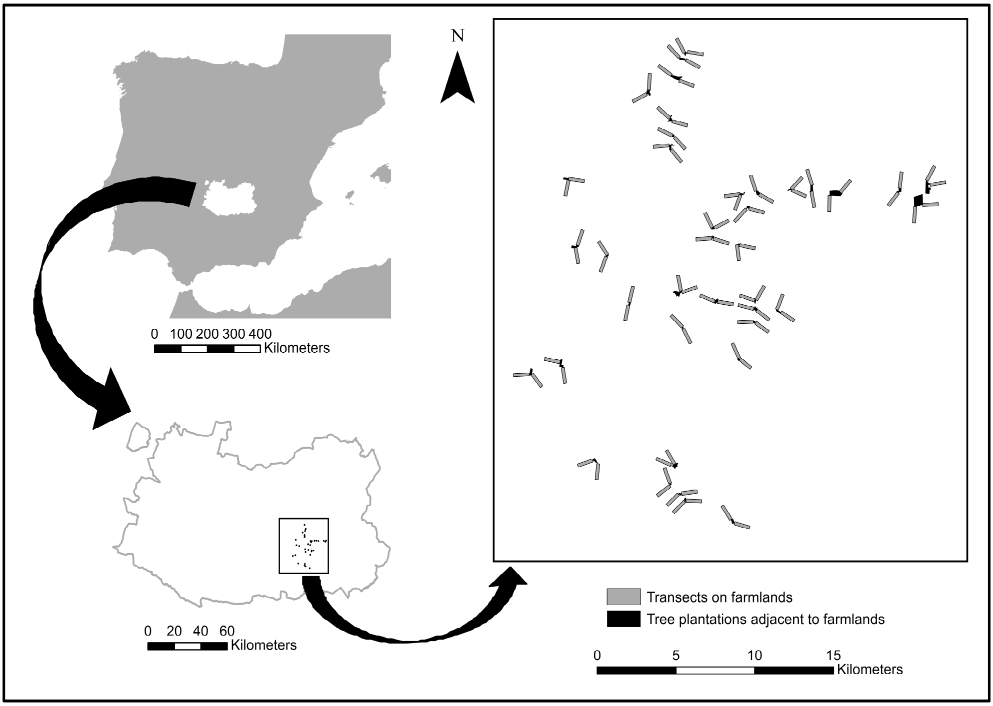

Figure 1. Location of the study area in central Spain within the Ciudad Real province and distribution of the young forest plantations (in black) and transects (grey) on adjacent cropland that were investigated in this study.

Selection of tree plantations for bird survey at adjacent farmland habitat

First, all tree plantations in the study area were located using both orto-photos (Geographic Information System of Farming Land, 2010; hereafter SigPac) and Google Earth®, and were later verified in the field. We found 99 tree plantations on former cropland that took place in 1992 or later. Tree plantations < 1 ha were directly discarded. In addition, a target tree plantation had to be placed at least 2-km away from another plantation in the transect direction to avoid that surveyed birds associated to open farmland adjacent to a given tree plantation were affected by another tree plantation. Following these criteria, we finally selected 40 tree plantations to assess bird community on farmland adjacent to tree plantations. We measured the area of every tree plantation (**Table 1**) using ArcGIS 10.0 (ESRI Inc.). As they are young, the tree canopy is little developed (mean tree cover = 38.8% ± 25.7%, mean tree height= 3.6 m ± 1.5 m, and mean dbh = 13.3 cm ± 6.5 cm).

Table 1. Mean, standard deviation (sd) and range (min / max) of area of tree plantations and land-use categories in 1-km x 200-m 80 transects on farmland habitat adjacent to the 40 tree plantations (two transects per plantation) that were surveyed in Central Spain.

	Mean	sd	Range	
Area of tree plantation (ha)	5.8	6.6	1.3	36.5
Cover of the tree layer (%)	38.8	25.7	1.7	100.0
Average pine height (m)	3.6	1.5	1.0	7.2
Average trunk diameter of pines (dbh cm)	13.3	6.5	4.0	33.2
Streams and rivers (% cover)	0.3	0.9	0.0	6.4
Roads and rural tracks (% cover)	1.7	1.6	0.0	7.7
Olive groves (% cover)	14.2	21.5	0.0	94.5
Scattered buildings (% cover)	0.3	1.0	0.0	7.3
Afforestation (% cover)	0.8	2.5	0.0	22.7
Semi-natural woodland (% cover)	0.1	0.8	0.0	9.5
Fruit and dried fruit groves (% cover)	0.2	2.1	0.0	26.6
Waste lands (% cover)	1.7	4.6	0.0	31.8
Pastures (% cover)	6.5	15.8	0.0	99.2
Dry herbaceous cropland (% cover)	40.4	32.8	0.0	100.0
Vineyards (% cover)	33.9	32.2	0.0	100.0

Bird survey

Species abundance and frequency was measured by means of census transects that were carried out in winter (January and February) and breeding season (April and May) of 2012 to assess the wintering and breeding bird communities, respectively. Census method consisted of transects of 1000-m length and 200-m width (Bibby *et al.*, 2000; Gregory, Gibbons & Donald, 2004); two outward transects from the tree plantation edge were surveyed at each tree plantation (80 transects in total). The two transects from each tree plantation spanned on different directions that were established a priori to meet the criterion used for selection of tree plantations (see above). The progression line of the surveyed transects was initiated at the tree plantation edge and considered a width of 100 m at each side; they were walked at an average speed of 2.5 km h-1. We noted and geo-localized the presence of every bird except those that were over-flying the census area (i.e., distance to the tree plantation edge and situation with respect to the transect progression line; **Table S1** in supplementary material). The two census-transects for each plantation and season were carried out in different days, one in the morning between sunrise and three hours later and one in the evening two hours before sunset. All censuses were conducted by the same well trained field ornithologist (JS S-O) on windless and rainless days.

The European endangered status of each species was obtained from BirdLife International (2004b) using the Species of European Conservation Concern (SPEC index) scores. This index uses four categories: SPEC 1 (global conservation concern), SPEC 2 (concentrated in Europe and with an unfavourable globally threatened or near threatened conservation status or data deficient), SPEC 3 (not concentrated in Europe but with an unfavourable conservation status), and Non-SPEC (favourable conservation status). We assigned a value of 4 to species that were included in the Non-SPEC category. Finally, we used a transformed SPEC index by subtracting the SPEC value of each surveyed species to 5 in order that species of highest conservation concern attained the greatest value (4), whereas the species of lowest conservation concern attained a value of 1 (de la Montaña, Rey Benayas & Carrascal, 2006). The average values of the transformed SPEC index were calculated by means of the recorded species in each transect (**Table S1** in supplementary material).

To have a reference of the avifauna that colonizes farmland habitat in the studied region, for comparison with our bird survey, we used (1) the species list (47 species) of the common farmland bird indicator for Southern Europe (European Bird Census Council 2010; http://www.birds.cz/pecbm/indik_lists.php?list_species=1&result_set=Publish2012&indik=E_C_SE_Fa) and the list (36 species) of common farmland bird index (Directorate-General for Agriculture and Rural Development 2012), and (2) the mean density of breeding species found at the habitat categories labelled as (a) dry arable lands, (b) vineyards, (c) olive groves, (d) agricultural mosaics with woody cultivations, and (e) pastures within the Mesomediterranean region of Central Spain obtained from Carrascal & Palomino (2008) (**Table S1** in supplementary material).

Land use types

We measured percentage of land use types in all 80 transects on farmland habitat where bird survey took place using ArcGIS 10.0 (ESRI Inc.). Land use types were identified by means of land use layers taken from SigPac (Geographic Information System of Farming Land, 2010) and verified in the field. We initially distinguished

21 land use types that were aggregated into the following eleven categories for statistical analyses according to their larger covers in the study region (i.e., avoiding those habitat categories of very low representativeness): streams and rivers, roads and rural tracks, olive groves, scattered buildings, afforestations, semi-natural woodland, fruit and dried fruit groves, waste lands, pastures, dry herbaceous cropland, and vineyard. The percentage of area occupied by each land use type across transects is shown in **Table 1**.

Statistical analyses

For statistical analyses we did not consider water birds (e.g., common sandpiper *Actitis hypolecus*, mallard *Anas plathyrhinchos*, and grey heron *Ardea cinerea*), aerial feeders (European bee-eater *Merops apiaster* and the *Hirundinidae* family species), and raptors. We only considered bird species that occurred at least in two segments along the surveyed 1-km transects, <400 m or close to tree plantations and 600-1000 m or away from tree plantations, leaving out from the analyses a 200-m wide transition zone. We run two sets of analyses separately, one for the winter and another for the breeding season.

Species richness and bird community SPEC score were analyzed by means of General Linear Models, with distance to tree plantation and plantation area as target predictors. Distance to tree plantation edge was treated as a dummy variable, i.e. 0 for close or <400 m vs. 1 for away or 600-1000 m. Area of tree plantation was included as a continuous covariate (in logarithm). As land use type may affect the abundance and frequency of species, we included in the models the cover of six land use categories with percentage higher than 1% as control covariates (namely, roads and rural tracks, olive groves, waste lands, pastures, dry herbaceous croplands and vineyards). Transect (n = 80) was the analytical unit for this analysis.

The occurrence of the commonest bird species (i.e., present in at least six of the 80 transects once getting rid of the 200-m wide transition zones) was analysed by means of Generalized Linear Models (i.e., logistic regression based on a Poisson distribution and the log-link function) for each season, using the same predictor variables than for species richness and SPEC scores (i.e., distance to plantations as a dummy variable, log-area of tree plantations, and cover of six land use categories).

The effect of distance to tree plantation on overall community composition –

referred to the abundance of the commonest bird species as defined above- was tested by means of a Semi-parametric Permutational Multivariate Analysis of Variance (hereafter PERMANOVA; Anderson, 2001). We used again the two defined distance groups (close vs. away) and tree plantation area and six land use categories as covariates.

GLMs and logistic regressions were carried out with Gretl (release 1.9.5, http://gretl.sourceforge.net/). Statistical significance of the predictor variables was calculated using quasi-ML standard errors. Robust estimations can provide for asymptotically valid statistical inference in models that are basically correctly specified, but in which the errors do not strictly follow canonical assumptions of statistical procedures. We also tested for homogeneity of slopes of plantation area in the close and away transect sectors in a posteriori regression analyses in all models. No interaction term distance*area was significant, so the effects of plantation area are generalizable across distance from the edge of the plantations. The PERMANOVA was performed with R statistical software (R Development Core Team, 2010) and the 'vegan' package (Oksanen *et al.*, 2011).

For visual inspection, we plotted for each species the mean abundance in the 80 1-km transects against the mean distance of occurrence at farmland habitat to tree plantation and the plantation area.

Results

Bird survey

We detected a total of 4218 individuals belonging to 59 species in winter and 1642 individuals belonging to 49 species in the breeding season at our 80 1-km transects (**Table S1** in supplementary material). Once getting rid of species not included in statistical analysis (i.e. water birds, aerial feeders, and raptors), we came up with 49 species in winter and 37 species in the breeding season. Twenty-one and 18 species in winter and breeding season, respectively, were present in ≥6 transects, which were represented by 3119 and 1096 individuals, respectively (**Table 2**).

Species richness, conservation status and community composition

Models revealed significant effects of distance to tree plantation edge on community species richness in winter (i.e., communities at >600 m from the plantations were ca. 30% richer in species that those at <400 m) but not in the breeding season (**Table 2**). The plantation area term did not have any effect on species richness in both seasons.

Table 2. Species richness, the transformed SPEC index related to conservation concern (1-safe to 4-highly threatened), and relative frequency of occurrence for species close to (<400 m) and away (>600 m) from forest plantation edges, in winter (A) and in the breeding season (B). Figures are mean ± sd. The regression coefficient and p-value of the effects of distance to plantation edge and plantation area (log-transformed) were obtained using general and generalized regression models that compare close vs. away (as a dummy variable: 0-close, 1-away) controlling for the effects of habitat type (see Methods for more details). Variables with at least one p-value significant at p<0.05 are bolded.

A. Winter	Close	Away	Distance		Area of plantation	
	Mean±SD	Mean±SD	Coeff.	p	Coeff.	p
Species richness	2.76±2.06	3.56±2.25	0.127	0.015	-0.045	0.570
Transformed SPEC index	1.39±0.68	1.48±0.64	0.054	0.337	-0.028	0.694
Alauda arvensis	0.15+0.36	0.15+0.36	0.064	0.775	-0.013	0.955
Alectoris rufa	0.14+0.34	0.14+0.34	-0.070	0.774	-0.792	**0.008**
Anthus pratensis	0.05+0.22	0.08+0.26	0.293	0.470	-0.614	0.192
Carduelis cannabina	0.21+0.41	0.31+0.46	0.312	0.118	-0.160	0.515
Carduelis carduelis	0.39+0.49	0.46+0.50	0.167	0.363	-0.150	0.542
Columba livia	0.05+0.22	0.13+0.33	0.599	**0.033**	-0.264	0.549
Columba palumbus	0.16+0.37	0.11+0.32	-0.360	0.145	0.371	0.286
Fringilla coelebs	0.20+0.40	0.26+0.44	0.102	0.632	-0.226	0.427
Galerida cristata	0.29+0.45	0.24+0.43	-0.041	0.825	-0.221	0.396
Lanius meridionalis	0.04+0.19	0.05+0.22	0.366	0.410	-0.659	0.264
Melanocorypha calandra	0.18+0.38	0.13+0.33	-0.248	0.283	0.252	0.441
Motacilla alba	0.06+0.24	0.08+0.26	0.346	0.321	-0.551	0.307
Parus major	0.08+0.26	0.03+0.16	-1.365	**0.030**	0.650	0.368
Phoenicurus ochruros	0.04+0.19	0.06+0.24	0.214	0.582	0.490	0.444
Phylloscopus collybita	0.08+0.26	0.13+0.33	0.172	0.572	0.482	0.201
Pica pica	0.16+0.37	0.36+0.48	0.552	**0.008**	0.352	0.213
Pterocles orientalis	0.00+0.00	0.09+0.28	10.616	**0.000**	0.618	0.387
Saxicola rubicola	0.06+0.24	0.03+0.16	-0.310	0.448	0.237	0.628
Serinus serinus	0.06+0.24	0.18+0.38	0.656	**0.017**	-0.157	0.617
Sturnus unicolor	0.08+0.26	0.13+0.33	0.354	0.151	-0.558	0.236
Sylvia atricapilla	0.05+0.22	0.03+0.16	-0.486	0.412	-0.214	0.796

B. Breeding season	Close	Away	Distance		Area of plantation	
	Mean±SD	Mean±SD	Coeff.	p	Coeff.	p
Species richness	3.20±1.88	3.01±1.87	0.000	0.996	-0.041	0.490
Transformed SPEC index	1.86±0.67	1.60±0.72	-0.116	**0.037**	0.011	0.879
Alauda arvensis	0.08+0.26	0.06+0.24	-0.143	0.662	0.210	0.547
Alectoris rufa	0.16+0.37	0.13+0.33	-0.136	0.586	-1.175	**0.008**
Burhinus oedicnemus	0.04+0.19	0.08+0.26	0.476	0.241	0.119	0.776
Carduelis cannabina	0.40+0.49	0.34+0.47	-0.133	0.451	-0.076	0.737
Carduelis carduelis	0.24+0.43	0.28+0.45	0.069	0.716	0.145	0.559
Columba livia	0.18+0.38	0.16+0.37	0.043	0.845	-0.254	0.383
Columba palumbus	0.36+0.48	0.34+0.47	-0.038	0.826	0.103	0.643
Emberiza calandra	0.11+0.32	0.05+0.22	-0.266	0.428	-0.741	0.100
Fringilla coelebs	0.04+0.19	0.06+0.24	0.324	0.410	-0.283	0.671
Galerida cristata	0.45+0.50	0.43+0.49	0.013	0.935	0.001	0.995
Melanocorypha calandra	0.13+0.33	0.09+0.28	-0.144	0.595	0.300	0.244
Passer domesticus	0.05+0.22	0.04+0.19	-0.02	0.956	-0.712	0.193
Pica pica	0.28+0.45	0.30+0.46	0.134	0.484	0.418	0.095
Pterocles alchata	0.10+0.30	0.13+0.33	0.162	0.538	0.026	0.937
Serinus serinus	0.04+0.19	0.04+0.19	0.116	0.811	0.470	0.465
Streptopelia decaocto	0.06+0.24	0.04+0.19	-0.145	0.717	-1.834	**0.005**
Sturnus unicolor	0.10+0.30	0.13+0.33	0.198	0.436	-1.213	**0.004**
Tetrax tetrax	0.16+0.37	0.10+0.30	-0.271	0.298	0.728	**0.023**

The assignment of the surveyed species to SPEC categories was the following: 32 species in the category Non-SPEC – least conservation concern –, 12 in the SPEC 3, nine in the SPEC 2, and two in the SPEC 1 – highest conservation concern (**Table S1** in supplementary material). Distance to tree plantation edge showed significant effects on the transformed SPEC index in breeding season (i.e., 19% higher index close to tree plantations), but not in winter. Again, the plantation area term did not have any effect on the transformed SPEC index in either season (**Table 2**).

Results of the PERMANOVA (**Table 3**) indicated that distance to tree plantation edge significantly affected overall community composition in the breeding season but not in the winter. Proportion of dry herbaceous cropland explained variation of community composition in both the winter and the breeding season and, additionally, proportion of olive groves explained variation in the winter.

Table 3. Effects of distance (close vs. away) and covariates (tree plantation area and land use categories) on variation in community composition in winter (A) and in the breeding season (B) according to a PERMANOVA. Variables with p-value significant at p<0.05 are bolded.

A. Winter	df	Sum of squares	R^2	F-statistic	p
Distance	1	0.412	0.00691	1.00816	0.459
Area of tree plantation (ha)	1	0.482	0.00809	1.1794	0.288
Roads and rural tracks (%)	1	0.418	0.00701	1.02241	0.422
Olive groves (%)	1	1.209	0.02027	2.95642	**0.001**
Waste lands (%)	1	0.479	0.00803	1.17072	0.260
Pastures (%)	1	0.306	0.00513	0.7478	0.781
Dry herbaceous croplands (%)	1	1.001	0.01678	2.44778	**0.003**
Vineyards (%)	1	0.139	0.00232	0.33901	0.998
Residuals	135	55.2	0.92547		
Total	143	59.645	1		

B. Breeding season	df	Sum of squares	R^2	F-statistic	p
Distance	1	6.902	0.1842	34.965	**0.001**
Area of tree plantation (ha)	1	0.284	0.00758	1.439	0.195
Roads and rural tracks (%)	1	0.256	0.00683	1.297	0.251
Olive groves (%)	1	0.208	0.00556	1.055	0.381
Waste lands (%)	1	0.044	0.00118	0.224	0.981
Pastures (%)	1	0.249	0.00664	1.26	0.270
Dry herbaceous croplands (%)	1	0.444	0.01186	2.251	**0.034**
Vineyards (%)	1	0.262	0.007	1.329	0.194
Residuals	146	28.818	0.76915		
Total	154	37.467	1		

Effects of distance to and area of tree plantations on species frequency

In winter, distance to tree plantation edge affected the frequency of occurrence of feral pigeon *Columba livia*, common magpie *Pica pica*, black-bellied sandgrouse *Pterocles orientalis*, and european serin *Serinus serinus*, which attained a higher frequency away from tree plantations (**Table 2A** and **Figure 2A**). However, the great tit *Parus major* had a higher frequency of occurrence at shorter distances from tree plantations. Larger tree plantations had a detrimental effect on the frequency of red-legged partridge *Alectoris rufa* (**Table 2A** and **Figure 2A**).

In the breeding season, distance from tree plantation did not exert any influence on species occurrence (**Table 2B** and **Figure 2B**). Tree plantation area had a detrimental effect on the frequency of occurrence of red-legged partridge, Eurasian collared dove *Streptopelia decaocto* and spotless starling *Sturnus unicolor*. However, little bustard *Tetrax tetrax* was more frequent near large plantations (**Table 2B** and **Figure 2B**).

Discussion

Overall, we found that young tree plantations established on former cropland in a Mediterranean mosaic located in central Spain had a detrimental effect on bird species that are characteristic of open farmland habitat (hypothesis 1), which was of relatively low intensity and different in the winter and the breeding season, and favoured very few bird species that are forest generalist or ubiquitous species (hypothesis 2).

Effects on open farmland bird species

Previous studies on the effects of tree plantations in open habitat bird species have mostly found negative effects, particularly for the most specialized and of more conservation concern (Shochat *et al.*, 2001; Santos *et al.*, 2006; Devictor, Julliard & Jiguet, 2008; Caplat & Fonderflick, 2009; Reino *et al.*, 2009, 2010, 2013; Butler *et al.*, 2010; Voříšek *et al.*, 2010; Fonderflick *et al.*, 2010; Morgado *et al.*, 2010; Méndez, Tella & Godoy, 2011). For instance, Fonderflick, Besnard & Martin (2013) found that the abundance of open-habitat birds decreased significantly in the vicinity of edges, this negative response extended within 150 m from the edge, and the effect was disproportionately higher in open-habitat species with high conservation concern. Accordingly, we found this detrimental effect for species richness and frequency of five common farmland species in winter (**Table 2A**) and differences in overall community composition in the breeding season (**Table 3B**). Four of these species are relatively large in size and two (red-legged partridge and black-bellied sandgrouse) are clearly linked to open farmland habitat and of conservation value at the European level (**Table S1**; BirdLife International, 2004b) and the regional level, where they present declining populations (SEO/Birdlife, 2013).

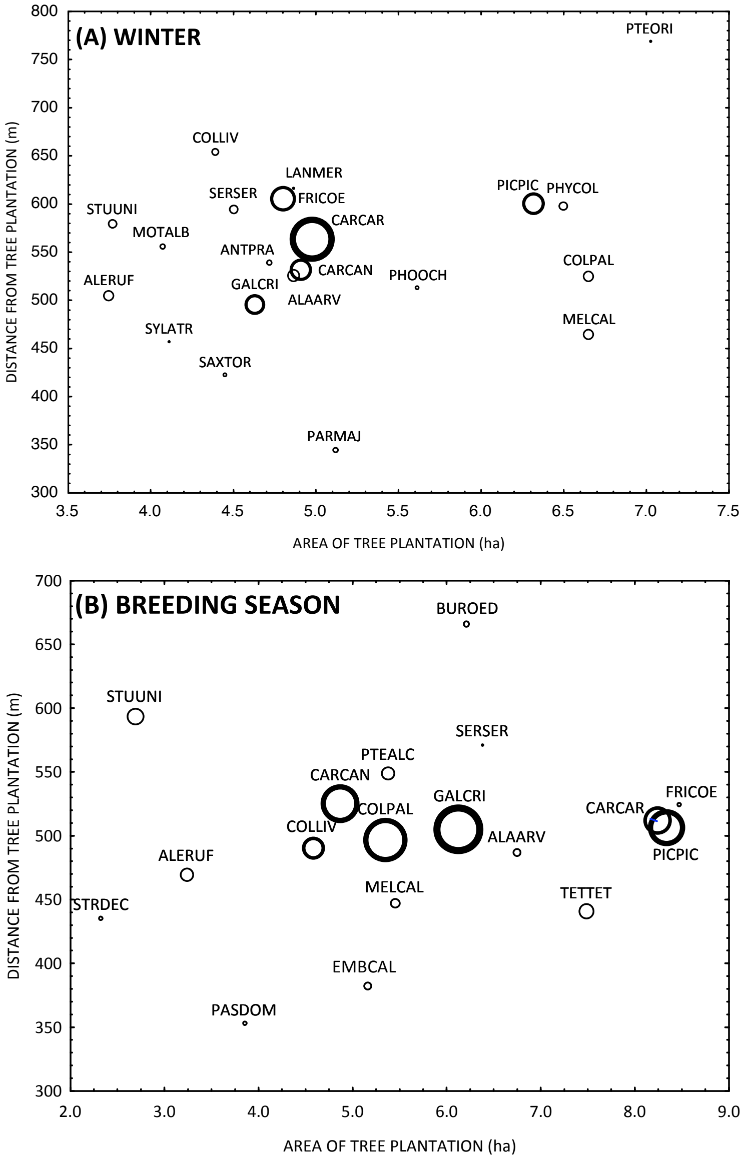

Figure 2. Plot of mean distance of occurrence at farmland habitat from tree plantation and the plantation area in the whole sample of 80 transects for each species in (A) winter and (B) breeding season. Frequency of occurrence is proportional to circle size. For more details on variability around means (i.e., coefficient of variation) see **Table S2**. Abbreviations are the following: ALAARUV: *Alauda arvensis*; ALERUF: *Alectoris rufa*; ANTPRA: *Anthus pratensis*; BUROED: *Burhinus oedicnemus*; CARCAN: *Carduelis cannabina*; CARCAR: *Carduelis carduelis*; COLLIV: *Columba livia*; COLPAL: *Columba palumbus*; EMBCAL: *Emberiza calandra*; FRICOE: *Fringilla coelebs*; GALCRI: *Galerida cristata*; LANMER: *Lanius meridionalis*; MELCAL: *Melanocorypha calandra;* MOTALB: *Motacilla alba;* PARMAJ: *Parus major;* PASDOM: *Passer domesticus*; PHOOCH: *Phoenicurus ochruros*; PHYCOL: *Phylloscopus collybita*; PICPIC: *Pica pica;* PTEALC: *Pterocles alchata*; PTEORI: *Pterocles orientalis*; SAXTOR: *Saxicola rubicola*; SERSER: *Serinus serinus*; STRDEC: *Streptopelia decaocto;* STUUNI: *Sturnus unicolor*; SYLATR: *Sylvia atricapilla* TETTET: *Tetrax tetrax*.

Contrary to our first hypothesis, conservation status of the bird assemblage in the breeding season was higher at close distance to the tree plantation edge and frequency of most common farmland species was not affected by distance to or area of tree plantations (**Table 2**). However, a few species that are included in one or both reference lists of common farmland birds, and that are present in our study region (Carrascal & Palomino 2008), were not detected by our surveys: *Anthus campestris, Calandrella brachydactyla, Galerida theklae, Lullula arborea, Oenanthe oenanthe, Sturnus vulgaris*, and *Sylvia communis*. Little bustard, a large and high conservation concern species, was associated to larger plantations in the breeding season in this study, but its abundance was twice away from than close to plantations (**Table S1**).

The small size of the plantations (5.8 ha in average) together with the little development of some of them (e.g. tree cover of 1.7%, **Table 1**) may produce detrimental effects only at very short distances from them (e.g. <150 m, Fonderflick *et al.*, 2013; Sánchez-Oliver *et al.*, 2013a). Further, these plantations may mirror remnants of natural or semi-natural woody vegetation such as woodland patches and hedgerows that may be even beneficial for some farmland bird species (e.g. buntings), as they offer opportunities for forage, refuge and breeding (Concepción & Díaz, 2010, 2011; Morgado *et al.*, 2010; Batáry *et al.*, 2012). Importantly, the hypothesized detrimental effect of the tree plantations seems to be diluted by the high heterogeneity of the landscape and the proportion of dry herbaceous cropland and olive groves (**Table 3**). In agreement, other studies have shown that landscape heterogeneity is a relevant factor affecting the occurrence and abundance of farmland birds (Morales, García & Arroyo, 2005; Batáry, Matthiesen & Tscharntke, 2010; Batáry *et al.*, 2011; Concepción & Díaz, 2011; Flohre *et al.*, 2011; Sánchez-Oliver *et al.*, 2013a).

Effects on forest generalist and ubiquitous species

In agreement with our hypothesis 2, great tit -a forest generalist species- was more frequent and abundant at shorter distances to tree plantation edges as tree plantations but not open farmland is the habitat of this species. Two ubiquitous species with increasing populations in the region, the Eurasian collared dove and spotless starling (SEO/Birdlife, 2013), were favoured by small plantations (Sánchez-Oliver, Rey Benayas & Carrascal, 2013b). However, we did not find that the frequency or abundance of species feeding or/and breeding in both forest and open habitat sharply increased near edges (positive edge response) as Fonderflick *et al.* (2013) did. Thus, species such as magpie, *Columba palumbus, Streptopelia turtur, Sylvia* spp., *Carduelis* spp., *Passer* spp. and *Lanius* spp. did not show a clear trend in relation to distance to and area of the tree plantations (**Table 1**). Again, we attribute this phenomenon to heterogeneity of the studied agricultural landscape and its high proportion of woody crops.

Concluding remarks

Young pine plantations established on former Mediterranean cropland exert an overall detrimental effect on open farmland bird species that seems to be diluted by the high heterogeneity of the landscape and the proportion of dry herbaceous cropland and olive groves. Thus, these tree plantations should not be favoured, and even be extirpated, in agricultural landscapes that are highly valuable for ground-nesting bird species and open farmland communities (Traba *et al.*, 2006; Butler *et al.*, 2010; Sanderson *et al.*, 2013). The pine plantations hardly favoured species that are forest generalist or ubiquitous species. We recommend long-term assessments of afforestation in agricultural landscapes to fully understand and, consequently, reduce its impacts on biodiversity, particularly on ground-nesting birds.

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Supplementary material

Table S1. List of all species surveyed at 80 1000 m long x 200 m wide transects located on farmland habitat adjacent to tree plantations in Central Spain. For each species, the following information is provided for the entire transect and for two segments of the transect (i.e. < 400 m or close to tree plantations and > 600 m or away for plantations): frequency (number of transects where the species occurred), local abundance (number of individuals km-2, mean ± sd), regional abundance at six farmland habitat types in the Mesomediterranean region in the breeding season (number of individuals km-2, mean ± sd; source: Carrascal and Palomino 2008), SPEC (source: BirdLife International 2004b), inclusion in the European farmland bird index (EFBI, + if included, source: Directorate-General for Agriculture and Rural Development 2012), and inclusion in the list of common farmland birds in Southern Europe (CFMSE, + if included, source: European Bird Census Council). The study area was not part of the distribution area of 10 species included in the CFBI list (*Ciconia ciconia, Corvus frugilegus, Emberiza citrinella, Emberiza melanocephala, Lanius collurio, Lanius minor, Limosa limosa, Perdix perdix, Saxicola rubetra* and *Sturnus vulgaris*) and six species included in the CFMSE list (the former but *C. frugilegus, E. citrinella, L. limosa* and *S. rubetra*). * indicates the species that were not considered for statistical analyses in this study.

Species	Winter						Breeding season						Regional abundance	SPEC	CFBI	CFMSE
	Entere		Close		Away		Entere		Close		Away					
	Freq	Abund	Freq	Abund	Freq	Abund	Freq	Abund	Freq	Abund	Freq	Abund				
*Accipiter gentilis**	1	5			1	5								Non		
*Actitis hypoleucos**	1	5	1	5										3		
Aegithalos caudatus	1	5			1	5								Non		
Alauda arvensis	28	32.9±45.0	12	25.8±28.8	12	35.8±51.1	13	9.6±6.6	6	6.7±2.6	5	9.0±6.5	2.5±1.7	3	+	+
Alectoris rufa	23	13.3±11.1	11	8.6±3.9	11	8.2±4.0	22	10.9±8.1	13	8.5±3.8	10	8.0±3.5	34.5±21.6	2		+
Anthus pratensis	13	81.9±121.5	4	153.8±168.5	6	60.0±90.3								Non	+	
*Apus apus**							12	19.2±13.6	6	15.0±11.4	5	20.0±12.7		Non		
*Aquila adalberti**	1	5			1	5								1		
*Athene noctua**							2	5.0±0.0			2	5.0±0.0		3		
Bubulcus ibis	1	5			1	5								Non		
Burhinus oedicnemus	1	5			1	5	10	7.0±3.5	3	5.0±0.0	6	6.7±2.6	2.0±1.8	3	+	+
*Buteo buteo**	3	5.0±0.0			3	5.0±0.0								Non		
*Calandrella rufescens**	1	20												3		
Carduelis cannabina	41	73.5±120.5	17	74.4±116.0	25	62.6±125.6	47	17.4±14.8	32	8.3±4.7	27	12.6±11.2	46.2±21.6	2	+	+
Carduelis carduelis	57	65.7±165.3	31	21.6±26.5	37	71.2±199.7	36	13.9±11.1	19	11.3±8.5	22	8.6±4.1	77.8±46.4	Non		+

Species	Winter						Breeding season						Regional abundance	SPEC	CFBI	CFMSE
	Entere		Close		Away		Entere		Close		Away					
	Freq	Abund	Freq	Abund	Freq	Abund	Freq	Abund	Freq	Abund	Freq	Abund				
Carduelis chloris	5	8.0±4.5	2	5.0±0.0	2	10.0±0.0	1	10			1	10	44.8±37.1	Non		+
Carduelis spinus	1	5			1	5								Non		
*Circus aeruginosus**	2	5.0±0.0	1	5	1	5	3	5.0±0.0	1	5	2	5.0±0.0		Non		+
Clamator glandarius							1	5			1	5	1.3±0.8	Non		
Columba livia	17	50.0±86.0	4	50.0±38.5	10	56.5±110.7	29	34.8±36.7	14	41.4±45.8	13	21.0±20.0	26.5±14.0	Non		+
Columba oenas	4	30.0±31.9	1	75	3	11.7±7.6	1	35	0		1	5	0.4±0.6	Non		
Columba palumbus	23	22.8±46.2	13	30.0±59.4	9	11.1±3.3	46	14.2±7.1	29	8.4±4.0	27	9.8±6.1	30.2±11.4	Non		
Corvus corone	1	60			1	60							0.8±0.3	Non	+	+
Coturnix coturnix	1	5	1	5			1	5			1	5	0.9±0.5	3		
Cyanistes caeruleus	2	7.5±3.5			2	7.5±3.5								Non		
*Delichon urbicum**							1	5	1	5			21.0±24.2	3		+
Emberiza calandra	2	7.5±3.5			1	5	13	8.8±4.6	9	8.3±5.0	4	6.3±2.5	57.4±33.3	2	+	+
Emberiza cirlus	2	82.5±95.5			2	82.5±95.5							2.4±4.4	1	+	+
Emberiza hortulana							1	10			1	10	0.2±0.3	2	+	+
Emberiza schoeniclus	1	15	1	15										Non		
Erithacus rubecula	4	5.0±0.0	2	5.0±0.0	1	5								Non		
*Falco naumanni**							2	12.5±10.6	1	5	1	5		1		
*Falco tinnunculus**	4		3	5.0±0.0	1	5	4	10.0±4.1	2	10.0±7.1	2	10.0±0.0	5.8±2.7	3	+	+
Fringilla coelebs	30	65.0±108.5	16	25.3±37.2	21	66.2±114.6	11	8.6±4.5	3	6.7±2.9	5	10.0±6.1	37.5±40.2	Non		
Galerida cristata	35	15.4±9.7	23	10.0±6.9	19	11.8±8.2	55	13.5±10.5	36	10.1±6.9	34	7.9±4.6	135.8±57.6	3	+	+
*Hirundo rustica**							*11*	*15.5±10.8*	*6*	*15.0±9.5*	*5*	*6.0±2.2*	*83.0±40.4*	*3*	+	+
Lanius meridionalis	*8*	*5.6±1.8*	*3*	*6.7±2.9*	*4*	*5.0±0.0*	*5*	*8.0±6.7*	*2*	*12.5±10.6*	*2*	*5.0±0.0*	*3.7±1.9*	*3*		
Lanius senator							*3*	*6.7±2.9*	*1*	*5*	*3*	*5.0±0.0*	*10.4±7.3*	*2*	+	+
Lophophanes cristatus	*26*	*39.0±83.7*					*1*	*5*			*1*	*5*		*2*		
Melanocorypha calandra			*14*	*32.9±66.0*	*10*	*31.0±48.8*	*17*	*9.7±7.0*	*10*	*8.0±3.5*	*7*	*7.9±5.7*	*28.6±39.4*	*3*	+	+
*Merops apiaster**							*5*	*10.0±5.0*			*5*	*10.0±5.0*	*19.2±9.8*	*3*		+
*Milvus migrans**							*2*	*7.5±3.5*	*1*	*10*	*1*	*5*		*3*		
Motacilla alba	13	38.5±72.3	5	10.0±6.1	6	70.0±101.1	*1*	*5*	*1*	*5*			*6.5±5.2*	*Non*		+
Oenanthe hispanica							*2*	*5.0±0.0*	*1*	*5*	*1*	*5*	*2.8±2.5*	*2*	+	+
Parus major	9	11.1±7.8	6	11.7±6.1	2	5.0±0.0	4	5.0±0.0	2	5.0±0.0	2	5.0±0.0	24.6±22.3	Non		

Species	Winter						Breeding season						Regional abundance	SPEC	CFBI	CFMSE
	Entere		Close		Away		Entere		Close		Away					
	Freq	Abund	Freq	Abund	Freq	Abund	Freq	Abund	Freq	Abund	Freq	Abund				
Passer domesticus	4	105.0±59.3	1	180	3	80.0±39.1	9	12.8±10.6	4	11.3±9.5	3	20.0±13.2	337:9±114.1	3		+
Passer hispaniolensis	1	10			1	10	1	10	1	10			9.5±12.2	Non		
Passer montanus	1	15					1	5	1	5			12.6±19.7	3	+	+
*Pernis apivorus**							3	45.0±69.3	1	120	1	5		Non		
Petronia petronia	4	6.3±2.5	1	5	2	7.5±3.5							5.9±4.8	Non	+	+
Phoenicurus ochruros	8	7.5±5.3	3	8.3±5.8	5	5.0±0.0								Non		
Phylloscopus collybita/ ibericus	20	6.8±4.9	6	6.7±4.1	10	6.0±3.2								Non		
Pica pica	41	15.7±12.7	13	11.5±8.5	29	11.9±9.3	44	14.1±10.0	22	9.1±6.1	24	9.4±7.0	31.6±17.8	Non		+
Pluvialis apricaria	1	65	1	65										Non		
Pterocles alchata	4	87.5±38.8	1	40	1	55	21	19.3±18.3	8	16.9±11.3	10	19.0±23.8		3		
Pterocles orientalis	7	10.7±4.5			7	10.7±4.5	4	12.5±5.0	3	10.0±0.0	1	20		3		
Saxicola torquata	9	7.8±3.6	5	7.0±2.7	2	5.0±0.0	1	5	1	5			11.6±6.2	Non	+	+
Serinus serinus	20	60.8±69.7	5	46.0±51.2	14	67.1±77.1	6	13.3±2.6	3	11.7±2.9	3	15.0±0.0	95.6±105.8	Non	+	+
Streptopelia decaocto	3	18.3±12.6	2	7.5±3.5			9	7.8±5.1	5	7.0±2.7	3	6.7±29	7.4±2.7	Non		
Streptopelia turtur							6	6.7±2.6	3	8.3±2.9	2	5.0±0.0	7.6±6.6	3	+	+
Sturnus unicolor	17	45.9±55.8	6	46.7±87.7	10	44.5±31.7	21	37.6±47.1	8	10.0±6.5	10	43.5±44.7	179.7±83.3	Non	+	+
Sylvia atricapilla	6	6.7±2.6	4	5.0±0.0	2	7.5±3.5								Non		
Sylvia melonocephala	1	5			1	5								Non		
Sylvia undata	1	5	1	5										2		
Tetrax tetrax	7	261.4±369.5	1	15	4	357.5±480.0	20	26.0±47.2	13	7.3±3.9	8	28.1±40.8		1		
Turdus merula	5	7.0±2.7	2	5.0±0.0	2	10.0±0.0	1	5			1	5	27.5±26.0	Non		
Turdus philomelos	6	8.3±4.1	2	5.0±0.0	3	11.7±2.9								Non		
Turdus viscivorus	1	5			1	5								Non		
Upupa epops	1	20	1	20.0±0.0			4	5.0±0.0	2	5.0±0.0	2	5.0±0.0	5.2±2.4	3	+	+
Vanellus vanellus	1	5					1		1	5				2	+	+

Table S2. Coefficient of variation (%) of distance of occurrence at farmland habitat from tree plantation and the area of nearby plantation in the whole sample of 80 transects for each species in the breeding season and in winter. See **Figure 2** for mean figures of each species in both seasons (--: Species with no data in that season).

Species	Abbreviation	Spring		Winter	
		Distance	Area	Distance	Area
Alauda arvensis	ALAARV	51.4	118.7	44.9	42.1
Alectoris rufa	ALERUF	62.1	78.5	57.3	51.3
Anthus pratensis	ANTPRA	--	--	53.3	56.4
Burhinus oedicnemus	BUROED	33.8	142.9	--	--
Carduelis cannabina	CARCAN	50.7	98.3	58.0	85.3
Carduelis carduelis	CARCAR	58.2	115.2	34.2	88.4
Columba livia	COLLIV	58.5	103.3	36.3	79.5
Columba palumbus	COLPAL	54.3	138.2	48.3	87.8
Emberiza calandra	EMBCAL	67.2	117.1	--	--
Fringilla coelebs	FRICOE	55.1	102.6	35.0	61.4
Galerida cristata	GALCRI	56.5	126.1	56.4	122.9
Lanius meridionalis	LANMER	49.1	43.3	48.8	76.7
Melanocorypha calandra	MELCAL	59.7	66.3	47.7	67.5
Motacilla alba	MOTALB	--	--	33.3	64.7
Parus major	PARMAJ	102.5	60.2	64.3	155.1
Passer domesticus	PASDOM	56.3	50.6	69.4	63.4
Phoenicurus ochruros	PHOOCH	--	--	52.5	70.4
Phylloscopus collybita	PHYCOL	--	--	44.9	140.3
Pica pica	PICPIC	47.9	111.5	42.7	126.9
Pterocles alchata	PTEALC	57.2	73.2	26.5	51.1
Pterocles orientalis	PTEORI	66.4	22.4	11.5	64.2
Saxicola rubicola	SAXTOR	--	--	37.1	80.1
Serinus serinus	SERSER	51.9	92.4	38.1	48.5
Streptopelia decaocto	STRDEC	71.0	33.9	30.3	9.7
Sturnus unicolor	STUUNI	35.5	55.3	48.1	85.6
Sylvia atricapilla	SYLATR	--	--	46.5	76.7
Tetrax tetrax	TETTET	39.3	69.3	18.5	54.3

Foto: Perdiz roja (*Alectoris rufa*)

Autor: Javi Cáceres

Capítulo 5

El hábitat local y el paisaje influyen en la alta depredación de nidos de aves en campos de cultivo mediterráneos reforestados

Este capítulo reproduce íntegramente el texto del siguiente manuscrito:

Sánchez-Oliver, J.S., Rey Benayas, J.M. & Carrascal, L.M. (2013). Local habitat and landscape influence high predation of bird nests of afforested Mediterranean cropland. *Agriculture, Ecosystems and Environment*. En revisión.

Resumen

Los programas de reforestación promovidos por la Política Agraria Comunitaria de la UE han contribuido a extender las plantaciones forestales en campos de cultivo. Sin embargo, estas reforestaciones pueden causar graves daños a las especies de hábitats abiertos, especialmente a las aves con alto valor de conservación. Investigamos la depredación de nidos artificiales de aves en plantaciones forestales jóvenes y campos de cultivo adyacentes a ellas en el centro de España. La tasa de depredación después de dos y tres semanas de exposición fue muy alta, tanto en las plantaciones forestales (95,6%) como en los hábitats agrícolas próximos (94,2%). La relación perímetro/superficie de las plantaciones y el desarrollo del dosel arbóreo redujeron la tasa de depredación, mientras que la abundancia de urracas (*Pica pica*) la aumentó dentro de las plantaciones. En hábitats agrícolas próximos a las reforestaciones, el área de la plantación más cercana aumentó la tasa de depredación, mientras que la distancia al borde de la plantación y la vegetación leñosa semi-natural la disminuyeron. Además, la tasa de depredación de nidos fue muy alta en paisajes homogéneos dominados por cultivos herbáceos y pastos con pequeñas cantidades de cultivos leñosos y sin vegetación leñosa semi-natural remanente. Concluimos que (i) las reforestaciones pequeñas y jóvenes no deberían ser favorecidas, e incluso deberían ser eliminadas, en paisajes agrícolas que tienen alto valor para las especies que nidifican en el suelo, (ii) la abundancia de urracas debería reducirse para incrementar la supervivencia de los nidos y (iii) los parches de vegetación leñosa semi-natural y las lindes deberían ser restauradas a escala de paisaje para minimizar el riesgo de depredación.

Palabras clave Nidos artificiales – Hábitat agrícola – Tipos de usos del suelo – Abundancia de urracas – Plantaciones de pinos

Local habitat and landscape influence high predation of bird nests on afforested Mediterranean cropland

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Abstract

Afforestation programs such as the one promoted by the EU Common Agrarian Policy have contributed to spread tree plantations on former cropland. Nevertheless these afforestations may cause severe damage to open habitat species, especially birds of high conservation value. We investigated predation of artificial bird nests at young tree plantations and at the open farmland habitat adjacent to the tree plantations in central Spain. Predation rates were very high at both tree plantations (95.6%) and open farmland habitat (94.2%) after two and three week exposure. Plantation edge/area ratio and development of the tree canopy decreased predation rate, while magpie (*Pica pica*) abundance increased predation rates within tree plantations. The area of nearby tree plantations (positive effect), distance to the tree plantation edge (negative effect), and semi-natural woody vegetation (negative effect) explained predation rates at open farmland habitat. Therefore, nest predation rate was very high in homogenous landscapes dominated by herbaceous crops and pastures with small amounts of woody crops and no remnants of semi-natural woody vegetation. We conclude that (i) small and young afforestations should not be introduced, and even be extirpated, in agricultural landscapes that are highly valuable for ground-nesting bird species, (ii) magpie abundance should be reduced to enhance nest survival, and (iii) patches of semi-natural woody vegetation and hedgerows should be

restored at the landscape scale to minimize predation risks.

Keywords Artificial nests – Farmland habitat – Land use types – Magpie abundance – Pine plantations

Introduction

A significant amount of abandoned cropland, low productive cropland and pastureland has been converted into tree plantations in the last few decades, and ca. 7% of the total forest land in the world are tree plantations at present (FAO, 2011). Different afforestation programs have contributed to the spread of such tree plantations at the regional level. Three major projects, namely Three Norths Shelter Forest Project, Grain-to-Green Project and Natural Forest Conservation Program, have afforested 28 million ha and 45 million ha in the 2000-2005 and 2005-2010 periods, respectively, in China (Cao, 2011; Cao *et al.*, 2011). Similarly, the Common Agricultural Policy (CAP) has favoured the conversion of farmland into tree plantations in the European Union since 1992 by means of a scheme of aid for forestry measures in agriculture (EEC Council Regulation No. 2080/92), which has resulted in the afforestation of > 8 million ha to date (European Commission, 2013a, 2013b). Further, afforested cropland is expected to increase in the near future in countries such as Spain due to subsidies to afforestation of extirpated vineyards (Spanish Agrarian Guarantee Fund, 2012). Whereas tree plantations pursue a number of environmental and social services such as soil retention and carbon sequestration (Rey Benayas *et al.*, 2007), they may have noticeable effects on biological communities, as it has been exemplarily shown with birds (Shochat, Abramsky & Pinshow, 2001; Santos *et al.*, 2006; Bremer & Farley, 2010; Felton *et al.*, 2010; Lindenmayer *et al.*, 2010; Rey Benayas, Galván & Carrascal, 2010).

Agro-ecosystems are important for maintenance of bird diversity in Europe, especially for species of conservation concern (BirdLife International, 2004). The Directorate-General for Agriculture and Rural Development (2012), using *the common farmland bird index* as "a barometer of change for the biodiversity of agricultural land in Europe", shows a decline in these bird populations of ca. 20% between 1990 and 2008 (see also, Gregory *et al.*, 2005; Butler *et al.*, 2010; Guerrero *et al.*, 2012). Cropland afforestations in southern Europe are mostly based on coniferous species such as *Pinus halepensis* and *P. pinaster*, and are an example of novel and hybrid ecosystems sensu Hobbs, Higgs & Harris (2009). These plantations may cause damage to open habitat species, especially birds, by replacing high quality open farmland habitat and increasing risk of predation (Díaz *et al.*, 1998; Cresswell, 2008; Reino *et al.*, 2009). Predation has both direct and indirect effects on bird populations (Batáry & Báldi, 2004), the latter related to the avoidance of use of habitats that are perceived as risky (Murcia, 1995) or fecundity reduction through sublethal trait-mediated effects (Bonnington, Gaston & Evans, 2013). Besides hindering the persistence of established ground-nesting bird populations, predation may impede the colonization of the new afforested habitat by bird species (Murcia, 1995; Lindenmayer & Fischer, 2006).

Tree plantations act as sources of generalist predators of various types, including rodents, lagomorphs, feral cats, dogs, and corvids (Andren, 1992; Pita *et al.*, 2009; Reino *et al.*, 2010; Suvorov *et al.*, 2012). These generalist predators usually have very low densities at treeless open habitats, but thrive in mosaic habitat landscapes where they exhibit an exploratory behaviour (Andren, 1992; Pita *et al.*, 2009; Reino *et al.*, 2010). Particularly, predation by corvids is enhanced in humanized landscapes where they attain high densities (Jokimaki, Huhta & Jokimäki, 2000; Newson *et al.*, 2010), and Suvorov *et al.* (2012) experimentally showed that the presence of magpie (*Pica pica*) nests increased predation rates on bird eggs. Accordingly, Castilla *et al.* (2007) attributed in part the relatively low predation on Red-legged partridge (*Alectoris rufa*) eggs at Mediterranean fallow fields to the low presence of magpies due to their capture by humans. Magpies are strongly attracted by trees in deforested landscapes for nesting, and this phenomenon is highly noticeable at relatively small and isolated tree plantations in Mediterranean cropland afforestations.

In this study we aim at investigating the predation of bird eggs set on artificial nests at young (< 20 yr) tree plantations established on former cropland and at the open habitat adjacent to such tree plantations in a farmland and woodland Mediterranean mosaic. We hypothesized that nest predation will be affected by both (1) the features of local breeding habitat and (2) the features of landscape – namely proportion of land use types- surrounding local habitat. At tree plantations, we predict that (i) a reduced area and a high edge-area ratio will favour permeability to predators and hence increase nest predation rates and (ii) magpie abundance and predation rate will be positively correlated. At open farmland habitat adjacent to tree plantations, we predict that predation rates will be influenced by (i) plantation area (positive effect), (ii) distance from plantation (negative) and (iii) magpie abundance (positive).

Our experimental study will allow explaining the risk of nest predation at Mediterranean landscapes that have been subjected to afforestation projects of former cropland, and provide hints for impact assessment and management of such projects at the local habitat and landscape scales.

Methods

Study area

Field work was carried out in afforested cropland and open farmland located in Campo de Montiel (La Mancha natural region, southern Spanish plateau, 38º41’53”N, 2º51’54”W, **Figure S1** in Supplemental Material). The study area spreads on ca. 440 km^2 with altitude ranging between 690 and 793 m a.s.l. The climate is continental Mediterranean with dry and hot summers and cold winters. Mean annual temperature and total annual precipitation in the area during the last 30 years were 13.7 °C and 390 mm, respectively (Agencia Española de Meteorología, 2012). These figures were 16.6°C and 359.9 mm in 2011 and 15.8ºC and 362.9 mm in 2012, when our nest predation experiments took place (Junta de Castilla-La Mancha, 2013).

The area is a representative mosaic of different crops, pastures and semi-natural or introduced woody vegetation that are characteristic of large areas in Mediterranean landscapes. Croplands were mostly occupied by herbaceous crops (wheat and barley) and permanent woody crops (olive groves and vineyards). Natural vegetation consisted of holm oak (*Quercus rotundifolia L.*) woodland and riparian forests that have been mostly extirpated from this region. Until 1992, woodland cover was restricted to open holm oak parklands, usually grazed by sheep and goats. Major land use changes in the last 20 years are the abandonment of herbaceous cropland and vineyard extirpation and their subsequent afforestation with the native Aleppo pine (*Pinus halepensis* Mill.) alone or mixed with holm oak and (*Retama sphaerocarpa* (L.) Boiss). These tree plantations are noticeably dominated by pines as they establish better and grow faster than the other planted species.

Selection of tree plantations for predation experiments

The constraints associated to each habitat type, namely tree plantations and open farmland adjacent to tree plantations, prevented homogeneous experimental designs and sampling methods, and consequently data from the different experiments will not be directly analysed together (see below). Thus, we run two independent experiments of bird nest predation, (1) at tree plantations and (2) on open farmland. First, all tree plantations in the study area were located using both orto-photos (Geographic Information System of Farming Land, 2010; hereafter SigPac) and Google Earth®, and were later verified in the field. We found 99 tree plantations on former cropland that took place in 1992 or later. Only tree plantations > 0.78 ha were selected for the predation experiments. In addition, a target tree plantation for the experiment on adjacent farmland had to be placed at least 2-km away from another plantation to avoid that experimental nests associated to a given tree plantation were affected by another tree plantation. Following these criteria, we finally selected 30 tree plantations for the experiment at tree plantations and 38 tree plantations for the experiment on farmland adjacent to the tree plantations, with 20 plantations that were used in both experiments (**Figure S1** in Supplemental Material).

Survey of magpie abundance

We recorded the abundance of magpie as a potential nest predator in the studied tree plantations and open farmland habitat adjacent to such plantations. At every tree plantation, magpies were surveyed using point-count stations (Bibby *et al.*, 2000) lasting 10 minutes. The point-counts were located at the centre of each tree plantation. All auditory and visual contacts were recorded, but only those within a 50 m radius (0.78 ha; **Figure S2** in Supplemental Material) were used in subsequent analyses, in order to increase the probability of detection. Every point-count station was surveyed by two censuses in different days in May 2011, one within the first 4 h in the morning and another in the afternoon beginning 3 h before sunset. We used the average of the two counts as a measure of magpie abundance. The same trained person conducted all the censuses (JSS-O) on nearly windless (wind speed <3 m s^{-1}) and rainless days.

The open farmland habitat adjacent to 38 tree plantations was also surveyed for magpie abundance by means of one line transect of 400-m length and 200-m width (**Figure S2** in Supplemental Material). Again, all censuses were conducted by the same well trained field ornithologist (JSS-O) on windless (wind speed < 3 m s-1) and rainless days. We have employed two different census methods for sampling magpie relative abundance according to the limitations imposed by the size of pine plantations, where transects were not possible due to their small area. Nevertheless, this is not a concern in this study as the aim is not to compare magpie abundance inside *vs* outside plantations, but to relate the relative abundance of magpies to nest predation within plantations and outside plantations, separately.

Mean magpie abundance at the 30 tree plantations was 1.37 birds per ha (sd = 1.87, range = 0-6.41). Magpie abundance averaged 0.11 birds ha^{-1} (sd = 0.18, range =0-0.63, n=38) at open farmland near tree plantations. Other corvid species were disregarded as key predators of artificial nests because they were very scarce in the study area (the Carrion Crow, *Corvus corone*, was detected at only one open farmland adjacent to tree plantations, and other species such as the Jay, *Garrulus glandarius*, or the Raven, *C. corax*, were not observed in the study area).

We also looked at other possible nest predators (e.g. carnivores –including feral cats and dogs-, rodents, hedgehogs, hares, and reptiles – Ocellated lizard, Montpellier snake and Ladder snake) during our field work related to nest predation experiments and local habitat features (see below).

Nest predation experiments

The two nest predation experiments used quail (*Coturnix coturnix*) eggs that were layed on artificial wicker nests (two eggs at each artificial nest). All eggs had the same origin (i.e., supplier), were washed and then dried at air temperature before being used for the field experiments (Vander Haegen & DeGraaf, 1996; Conner & Perkins, 2003; Piper & Catterall, 2004), and were handled with gloved hands to minimize human scent (Whelan *et al.*, 1994).

Experiment 1.- Predation at the tree plantations. This experiment was run at 30 plantations in the spring of 2011, which averaged 5.6 ± 7.2 ha and ranged between 1.5 and 36.5 ha. The artificial nests with two quail eggs each were placed at two different positions (i.e. one nest on the ground and another nest on pine branches) at 25-m intervals along an *a priori* line spanning from the edge (0 m) to the centre of the plantation (**Figure S2** in Supplemental Material), in May 22-25. The height above the ground for those nests located on branches was estimated by means of a measuring tape. The minimum distance covered by the line over which the artificial nests were placed was 50 m (i.e. 3 x 2 nests located at 0, 25, and 50 m from the plantation edge), whereas the maximum length from the plantation edge was 225 m that included ten nest locations (average was 70.8 m and sd = 38.9). Total sample size was 230 nests, 115 located on the ground and 115 located on branches. We visited the nests in two occasions, 7-9 days (May 31 and June 1) and 15-18 days after they were placed (June 9-11), counting the number of eggs that had been removed. Artificial nests were not checked more often in order to reduce the effect of the observer on predation and to preserve nest concealment (e.g., Major and Kendal, 1996). We also annotated the likely type of predator (magpie, rodents or unknown) and incidences such as harm to eggs due to trampling by cattle.

Experiment 2.- Predation on open farmland adjacent to tree plantations. This experiment was run at 38 plantations in the spring of 2012. Each artificial wicker nest was baited with two fresh quail eggs (see above) and was placed on the ground along an *a priori* 300-m line spanning at 25-m intervals from the plantation edge (0 m) until 150 m away from such edge, and then at 50-m intervals until 300 m (i.e., nine nests at 0, 25, 50, 75, 100, 150, 200, 250, and 300 m; **Figure S2** in Supplemental Material). The artificial nests were placed on May 4-9. Total sample size was 342 nests. We took note of the habitat type where each nest was situated, considering five habitat categories (olive groves, vineyard, abandoned cropland and pastures, semi-natural woody vegetation, and dry herbaceous cropland). We checked the nests for egg predation in two occasions (in May 15-22 and in May 27-June 1, 11-14 days and 21-23 days after the nests were placed), following the same protocol presented in Experiment 1.

The artificial nests within plantations and in open fields near plantations were not placed on the same date due to limitations inherent to the organization of the field work, which included a number of tasks, and considering the timing of agricultural activities in the study area (e.g., ploughing). Nevertheless, the data for the two experiments were analysed separately and were never directly compared.

In both experiments, we considered an artificial nest as predated when the eggs were either absent or damaged, excluding from analyses those artificial nests that were ploughed or trampled (42 and 7, respectively, on open farmland and neither at tree plantations). Sources of predation could not be distinguished for the eggs that were removed from the artificial nests which, in turn, were the most. Nevertheless, unidentified predation events were probably attributable to small corvids (Schaefer, 2004) such as magpies considering their ability to store large items of food and to steal and remove eggs from nests (Henty, 1975; Groom, 1993; Perrins, 1998). We were able to distinguish predation by rodents (by their characteristic bites and, sometimes, faeces) and by corvids (by their characteristic pecks) from some fresh egg remains, whereas for the largest part of predated eggs with fresh remains we could not distinguish the source of predation. However, this issue is not a problem for the aims of this study since we are interested in the effects of tree plantations on overall predation risk rather than in the identification of predator assemblages.

Local habitat and landscape features

In each of the 46 tree plantations where experiments 1 and 2 took place, we characterized variables related to vegetation structure, area, edge/area ratio, and landscape surrounding the tree plantation. Vegetation structure at each surveyed plantation was characterized in one 25-m radius plot (**Figure S2** in Supplemental Material). We directly measured or estimated by eye, after previous training, the following structural features of the vegetation: percentage cover of chamaephytes, shrubs and trees, average height of chamaephytes, shrubs and trees, and number of trunks <5, 5-10, 10–20, 20–40 and >40cm in diameter at breast height (dbh). Additionally, we estimated percentage cover of herbs and bare soil and measured the average height of the herb layer in one concentric 10-m radius plots within the 25-m radius plot (**Figure S2** in Supplemental Material) due to perceptual limitations when carrying out visual estimations. Vegetation structure was sampled by the same observer (JSS-O) to avoid inter-personal bias in vegetation measurements. We also measured area and edge/area ratio using ArcGIS 10.0 (ESRI Inc.). Edge/area ratio was calculated as the quotient between the length of the edge (in m) and the square root of the plantation area (in m^2). The tree plantation and vegetation structure variables are reported in **Table S1** in Supplemental Material.

Land use types were identified by means of land use layers taken from SigPac (see source above) and were analysed with ArcGIS 10.0. We distinguished 14 land use types: streams, rivers and lagoons, roads and rural tracks, urban areas and scattered buildings, semi-natural woodland, dried-fruit orchards, orchards, waste lands, olive grove, pastures with scattered trees, scrubland, pasture land, dry herbaceous cropland, vineyard, and vineyard with olive trees. To characterize landscape surrounding the tree plantations for Experiment 1, the percentage of area of each land use types was obtained in 1-km buffer-rings centred at each forest plantation (**Figure S2** in Supplemental Material). To characterize landscape for Experiment 2, the proportion of land use types was measured as above at 600 m x 600 m squares that included the 300-m transects in farmland habitat were the artificial nests were set (**Figure S2** in Supplemental Material). Landscape characteristics associated to both predation experiments are reported in **Table S1** in Supplemental Material.

Statistical analysis

The two experiments of nest predation were analysed independently. We used predation incidence obtained from the first checking date as most artificial nests were predated within the first 7-14 days after they were placed on the field (see Results).

Experiment 1.- For predation within tree plantations, the plantation was the sample unit (n= 30) because preliminary analyses that considered the individual artificial nests as sample units indicated a highly contagious predation pattern according to the Moran's index ($p<0.001$; using SAM software, Rangel, Diniz-Filho & Bini, 2010). The spatial auto-correlation analysis performed on the 30 plantations using a lagged predictor model for percentage of predation (response variable) against the predictor variables (see below) resulted in a non-significant spatial effect ($p>0.12$ for five distance classes).

We performed two different principal components analyses (PCA), one on vegetation structure variables within tree plantations and another on land use types surrounding the plantations, to obtain synthetic and independent environmental gradients that may affect nest predation. The first component on vegetation structure variables within tree plantations (VPC1; 51.2% of total variance) defined an increasing gradient of development of the tree canopy, as it opposed tree cover (factor loading = 0.897), tree height (0.816) and number of trunks >5 cm in dbh (0.852) to shrub height (-0.724) and cover (-0.523) and herb height (-0.656). The second component on vegetation structure variables (VPC2; 20.1% of the total variance) was associated to the development of the shrub layer; it opposed shrub cover (0.727) and height (0.602) to herb cover (-0.611). The first component on land use around tree plantations (LPC1; 36.3% of the total variance) opposed olive groves (0.964) and semi-natural woodland (0.718) to roads and rural tracks (-0.842), vineyard (-0.766) and dry herbaceous cropland (-0.637).

We used a generalized linear model based on a Poisson distribution (with the log-link function) at tree plantations because the response variable was the number of predated nests, i.e. a quantitative and continuous variable, with the total number of artificial nests placed at each plantation as the offset. This model was applied to analyse the effects of eight predictor variables, namely tree plantation area (log-transformed), plantation edge/area ratio, average distance of artificial nests to the plantation edge, mean height of artificial nests, magpie abundance, two components related to vegetation structure (VPC1 and VPC2), and one component (LPC1) related to landscape. Statistical significance of the standardized regression coefficients of the predictor variables was estimated using a robust approach with quasi-ML standard errors (Lindsey, 2004).

Experiment 2.- For predation on open farmland habitat adjacent to tree plantations, the artificial nest was the sample unit because the land use type at which artificial nests where placed varied, and we were interested in testing the effect of distance to plantation edge. Moreover, the individual artificial nests as sample units indicated a very low contagious predation pattern according to the Moran's index ($p>0.2$; using SAM software, Rangel *et al.*, 2010). The number of artificial nests used for the statistical analyses was n = 259 as, out of the 342 nests placed in total for the experiment, 34 nests were not found, seven were trampled, and 42 were located on cropland fields that were ploughed. All artificial nests at two out of the 38 tree plantations that were initially selected for Experiment 2 were lost due to ploughing or trampling.

For this experiment we carried out another PCA on land use type categories measured at 600 m x 600 m squares. The first component (14.8% of the total variance) opposed semi-natural woodland (0.644) and pastures with scattered trees (0.626) to dry herbaceous cropland (-0.715).

We used a logistic linear model (binomial errors and logit link function) to analyse nest predation (0 *vs.* 1, i.e. non-predated *vs.* predated, a binary response variable). Predictor variables were: tree plantation area (log-transformed), distance of artificial nests to the plantation edge, magpie abundance, average tree height of the nearest plantation, the component related to landscape features, and habitat type where the artificial nest was placed (five habitat categories: olive groves, vineyard, abandoned cropland and pastures, semi-natural woody vegetation, and dry herbaceous cropland). The continuous predictor variables were standardized in order to obtain standardized regression coefficients, and the habitat category factor entered the model as dummy variables. Geographical position of nests was considered in the analysis to control for spatial non-independence (i.e., auto-correlation). The influence of the spatial location and proximity of nests was included in the logistic regression by means of a two-order polynomial of latitude and longitude, thus performing a trend surface analysis (Legendre, 1993).

We looked at the correlation among the independent variables of our models. Most correlations were not significant. Particularly, the shared variance between magpie abundance and other explanatory variables was usually very low (as measured by the coefficient of determination R^2): (a) Within tree plantations: log area 0.03; edge/area ratio <0.001; PC1 vegetation 0.14; PC2 vegetation 0.04; and PC1 land use 0.08; (b) On open farmland habitat: log area 0.09; pine height <0.001; and PC1 land use 0.03. Anyway, collinearity is not an important concern in our analyses as we only considered partial effects by means of multiple regressions.

GRETL software (Cottrell & Lucchetti, 2007) was used for generalized linear models. The principal components analyses were carried out using Statistica 10 (StatSoft, 2011).

Results

Predation rates

Overall predation rates were very high at both the tree plantations and adjacent open farmland (**Figure 1**). Most of the predated artificial nests were observed by the first counting, one to two weeks after being set. Only 4.4% and 5.8% of artificial nests at tree plantations and on open farmland habitat, respectively, were left un-predated two to three weeks after the start of the experiment (**Figure 1**).

All artificial nests at 12 (40%) tree plantations were predated by the first counting, and all artificial nests were left un-predated at only one tree plantation. On open farmland habitat, all artificial nests were predated in 21 (58.3%) transects by the first counting. The maximum number of artificial nests left un-predated in a transect by the first counting was 85.71%.

Of the total nests, 74.2% at tree plantations and 79.2% on open farmland were removed and, consequently, their source of predation is unknown. Predation by rodents at tree plantations and on open farmland were, respectively, 1.7% and 2.3%, whereas 5.2% and 6.9%, respectively, showed evidence of predation by corvids, namely magpie as the nearly exclusive corvid species present around and in plantations. Therefore, sources and levels of predation were very similar inside and outside the tree plantations.

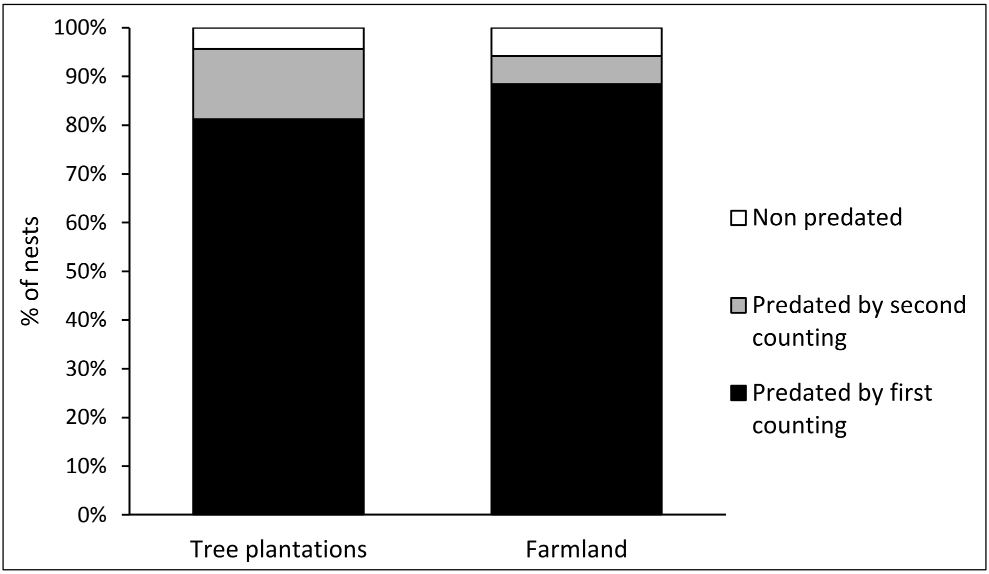

Figure 1. Percentage of predated artificial nests by the first counting and by the second counting and of non predated nests at tree plantations (Experiment 1) and on adjacent open farmland habitat (Experiment 2).

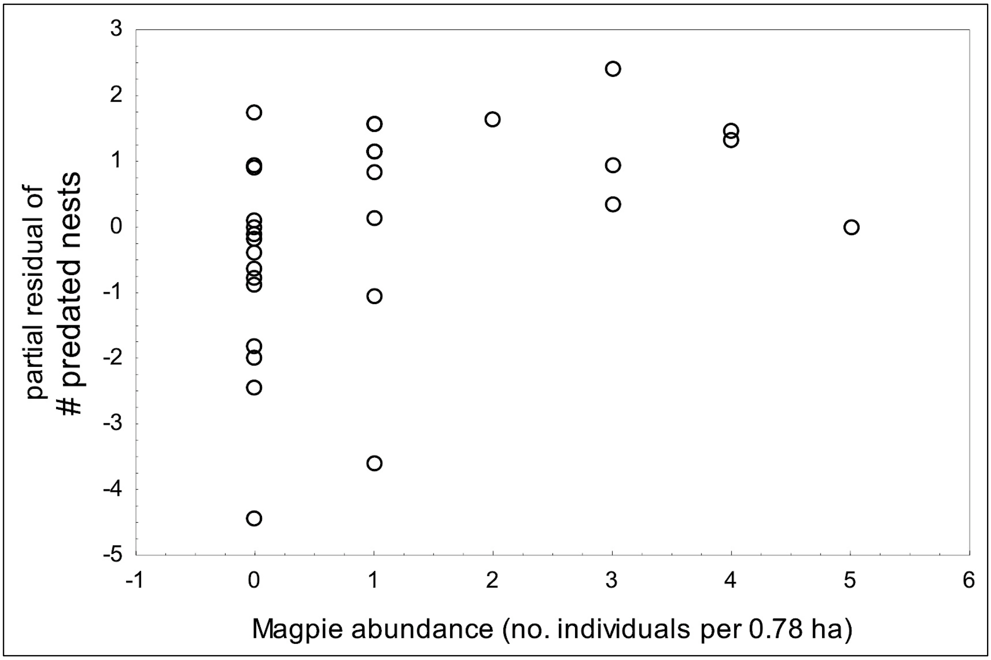

Figure 2. Partial residual plot of the influence of magpie abundance on predation intensity of nests at 30 tree plantations on former cropland. The residual plot shows the relationship with magpie abundance given that the other independent variables are also in the model, therefore partialling out their effects (see **Table 1** for more details).

Table 1. Results of the generalized linear model (Poisson distribution with log-link function) analyzing the effects of predictor variables on the number of predated artificial nests at 30 tree plantations on former cropland. p: statistical significance was estimated using a robust approach with quasi-ML standard errors. Significant predictor variables at p<0.05 are bolded. Beta: standardized partial regression coefficients.

	Beta	Se	Z	p
Area (ha; log-transformed)	0.153	0.089	1.720	0.086
Edge/area ratio	**-0.164**	**0.066**	**-2.475**	**0.013**
Distance to edge (m)	-0.030	0.082	-0.370	0.712
Mean height of nest (m)	0.033	0.042	0.773	0.440
Magpie abundance (no. individuals)	**0.163**	**0.057**	**2.861**	**0.004**
PC1 Vegetation structure	**-0.095**	**0.045**	**-2.108**	**0.035**
PC2 Vegetation structure	0.073	0.040	1.831	0.067
PC1 Land use types	**-0.152**	**0.066**	**-2.291**	**0.022**

Table 2. Results of logistic regression analysis showing the effects of predictor variables on predation of artificial nests on open farmland adjacent to 36 tree plantations on former cropland. p: statistical significance was estimated using a robust approach with quasi-ML standard errors. Significant predictor variables at P<0.05 are bolded. Beta: standardized partial regression coefficients for continuous predictors and standard coefficients for the dummy variables (0,1) built with the levels of the factor habitat types.

	Beta	SE	p
Standardized latitude (X) coordinate	1.653	0.89	0.063
Standardized longitude (Y) coordinate	-0.563	0.474	0.235
X^2	-0.226	0.664	0.734
Y^2	-0.305	0.467	0.514
X*Y	1.777	1.164	0.127
Area (ha; log-transformed)	**1.503**	**0.529**	**0.004**
Distance to edge (m)	**-0.602**	**0.255**	**0.018**
Magpie abundance (no. individuals)	-0.236	0.386	0.540
Average pine height (m)	0.021	0.239	0.929
PC1 Land use	**-1.188**	**0.484**	**0.014**
Habitat types where nests were placed:			**<0.001**
Olive groves	0.357	0.571	0.531
Vineyard	1.174	0.868	0.176
Abandoned cropland and pastures	0.946	0.537	0.078
Semi-natural woody vegetation	**-2.097**	**0.621**	**0.001**
Dry herbaceous cropland	0.838	0.739	0.257

Nest predation at tree plantations

Multiple regression analysis of the number of predated nests at tree plantations revealed statistically significant effects of edge/area ratio (negative effect on predation), magpie abundance (positive effect; **Figure 2**), development of the tree canopy (first PC of vegetation structure variables, VPC1, negative effect), and relative amount of tree crops and woodland in the landscape (first PC of land use type variables, LPC1, negative effect; **Table 1**). Tree plantation area, mean distance of artificial nests to the plantation edge, and mean height of the artificial nests above the ground did not show any significant effect on nest predation.

Nest predation on open farmland adjacent to tree plantations

Logistic regression analysis of nest predation on open farmland resulted in four statistically significant explanatory variables after controlling for geographic position of nest location (**Table 2**): area of the nearby tree plantation (positive effect on predation), distance to the tree plantation edge (negative effect), relative amount of tree crops and woodland in the landscape (first PC of land use variables, negative effect), and the habitat type semi-natural woody vegetation where the nest was placed (negative effect). Magpie abundance and mean height of nearby tree plantations did not show any significant effect on nest predation.

The area of the tree plantations for predated (n = 229) and non-predated (n = 30) artificial nests was 6.3±7.2 ha and 3.2±1.4 ha, respectively (log- transformed: 1.5±0.8 and 1.1±0.5). Predated and non-predated artificial nests were on average at a distance of 120.7±96.1 m and 140.0±111.9 from the tree plantations, respectively, and the modal values corresponded to a distance of 50 m for predated nests and 300 m for non-predated nests.

Discussion

Overall, we found that predation of artificial bird nests at young tree plantations established on former cropland and at adjacent open farmland habitat in a Mediterranean mosaic located in central Spain was (1) very high at both habitats, (2) influenced by local habitat features, and (3) influenced by landscape context. However, we obtained different results for specific variables that were hypothesized to affect predation rates inside and outside the investigated tree plantations (i.e. area, edge/area ratio, distance to edge, and magpie abundance).

Extraordinarily high predation rates

Nest predation was very high and quick at both the tree plantations and adjacent open farmland habitat (81.2% and 88.4%, respectively, in less than two weeks after the start of our experiments). These rates are among the highest reported in the scientific literature (data and references in **Table 3**). Previous published figures of nest predation rates at tree plantations average 59.5% with a range of 23-94% (**Table 3a**). Similarly, nest predation rates for natural forest fragments are usually high (mean = 66.4%, range = 38.9-88.0%; **Table 3b**) but lower than in our tree plantations (95.6%). Other studies that have assessed nest predation rates at open habitat adjacent to tree plantations or natural forest fragments reported figures that average 60.0% (range = 13.7-100%; **Table 3c**), which are substantially lower than our 94.2% predation rate. In an experiment that used eggs of Red-legged partridge located at Holm oak woodland patches in Central Spain, Castilla *et al.* (2007) reported a predation rate of 38.9% after a 2-week exposure.

We attribute the extraordinarily high predation rates in our study to the following three phenomena. First, our tree plantations were overall very small (mean size of 5.7±6.7 ha), which make nests easily accessible to predators in general even at the largest plantations (Ford *et al.*, 2001; Chalfoun, Thompson III & Ratnaswamy, 2002). Secondly, they were located in an agricultural and highly humanized landscape, which may favor predation by a number of animals such as rodents, hares, feral cats and dogs (Danielson, DeGraaf & Fuller, 1997; Jokimäki *et al.*, 2005; Pangau-Adam, Waltert & Mühlenberg, 2006). And third, they were a very attractive habitat for magpies, a documented powerful nest predator (Andren, 1992; Roos, 2004; Suvorov *et al.*, 2012) that was particularly abundant in our study area (see also Sánchez-Oliver, Rey Benayas & Carrascal, 2013).

Factors affecting predation rates

Nest predation at tree plantations increased with a higher abundance of magpies, a lower edge-area ratio and a lower development of the tree layer, whereas nest predation on open farmland habitat was higher if nearby tree plantations were of large area and nests were located at closer distances from the plantations.

Predator identification in our experiments was relatively unsuccessful as the proportion of eggs that disappeared was high (>74%) and, unfortunately, egg shell observation did not provide enough information to determine the main sources of predation. However, we detected a high correlation between nest predation rates

Table 3. Review of nest predation rates at (a) tree plantations, (b) forest fragments and (c) open habitat adjacent to tree plantations or forest fragments. The mean and range of predation rates and the mean ± sd of exposure days for the three habitat types (i.e. a, b and c) have been calculated by the authors of this study on the basis of the referred studies.

Habitat type	Landscape context	Mean (range) predation rate (%)	Mean no. days (±sd)	References
a) Tree plantations		59.5 (23.0-94.0)	12±2	
Conifer plantations	Sub-boreal forest	83.7 (64.7-94)	10	Pedersen et al., 2009
		23.0	14	Vander Haegen and DeGraaf, 1996
		41.2 (36.7-45.8)	13	Carignan and Villard, 2002
b) Forest fragments		66.4 (38.9-88.0)	11±4	
Forest fragments	Boreal agricultural	88.0	7	Andren, 1992
Oak forest fragments	Mediterranean agricultural	87.5	8	Santos and Tellería, 1992
		38.9	14	Castilla et al., 2007
Fagus forest fragments	Eurosiberian agricultural	41.7	14	Ludwig et al., 2012
Cloud forest fragments	Andean agricultural	48.9	15	Arango-Vélez and Kattan, 1997
Rainforest fragments	Tropical pastures	71.9	9	Estrada et al., 2002
c) Open habitat adjacent to tree plantation or forest fragments		60.0 (13.7-100)	10±3	
Forest fragments	Boreal agricultural	99.0	7	Andren, 1992
		41.0	14	Vander Haegen and DeGraaf, 1996
Tree plantations and Oak forest fragments	Mediterranean agricultural	49.0	15	Reino et al., 2010
		13.7	14	Castilla et al., 2007
		50.0	8	Santos and Tellería, 1992
Fallow	Template forest	86.5	14	Conner and Perkins, 2003
Rainforest fragments	Pastures with tropical rainforest remnant	79.0	9	Estrada et al., 2002
Clearing	Turkey Oak forest	24.0	7	Purger et al., 2004

and magpie abundance in tree plantations only, which points to relevance of nest predation caused by magpies but not the relative importance of magpies with regards to other predators. Magpies have a high capability of exploring relatively new habitats and are prone to nesting in the most developed plantations (> 3 m in height) that we surveyed in our mostly deforested study area. Andren (1992) found predation rates of bird nests in forest fragments by this corvid that ranged between 7.2% and 35.7%. As most of the studied tree plantations are of a rectangular shape, low edge-area ratios mean larger plantations, which may function as a refuge habitat and harbour a higher abundance of magpies and other generalist predators of bird nests such as domestic carnivores (Virgós, Tellería & Santos, 2002; Barea-Azcón *et al.*, 2006; Pita *et al.*, 2009; Fandos, Fernández-López & Tellería, 2012). The higher predation rates at tree plantations with lower tree development may be explained by the facts that these plantations are newer habitats that call more the attention of exploring predators (Virgós *et al.*, 2002) than older plantations and, additionally, they are more open and thus artificial nests are more visible (Suvorov *et al.*, 2012).

The small size and homogeneity of the studied tree plantations and the extraordinarily high predation rates explain why distance to edge and average height above the ground of the nests did not have any effect on predation rates at tree plantations. However, a shorter distance to edge of the tree plantation may enhance predation on the open farmland habitat because nests are closer to the source of predators (Batáry & Báldi, 2004; Reino *et al.*, 2010) such as magpies, which in this study attained an abundance four times higher near the plantations (<150 m) than away from them (>150 m). This figure together with lack of correlation between magpie abundance and nest predation on open farmland makes unclear if magpies are or not a major predator in open habitats, an issue that should be tested by a before/after complementary experiment (i.e., measuring predation rates before and after magpie removal, and compared the results with those from reference sites without magpie removal). We also found that nests placed on patches of semi-natural woody vegetation, namely hedgerows and *Q. rotundifolia* remnants, exhibited relatively low predation rates, and reinforces the importance of this vegetation for conservation of ground-nesting birds in vast open farmed fields (Santos *et al.*, 2006; Ludwig *et al.*, 2012).

Finally, we found significant landscape effects on nest predation at both the tree plantations and the surrounding open farmland habitat. Tree plantations favour nest predation in landscapes with higher proportion of herbaceous crops and pastures and lower proportion of woody crops and semi-natural woodlands. Other studies have found relationships between landscape context and nest predation rates (Huhta, Mappes & Jokimaki, 1996; Bayne & Hobson, 1997). In our study, we explain this result because tree plantations in open, deforested, and homogenous landscapes are better attractors and refuges of predators than tree plantations in more heterogeneous landscapes where there is more availability of habitat with trees (e.g. Andren, 1992), in accordance with lower predation rates on our artificial nests placed on patches of semi-natural woody vegetation. Deforested landscapes with a high proportion of herbaceous crops favour also the abundance of lagomorphs, which can predate on eggs (Reino *et al.*, 2010).

Concluding remarks

The use of artificial nests and clutches to test predation rates is controversial due to factors that are not controlled with respect to real nests, and several studies have demonstrated that artificial clutches do not estimate nest predation rates on natural nests precisely (Burke *et al.*, 2004; Faaborg, 2004; Thompson & Burhans, 2004; Villard & Part, 2004). Also, nest predation is only one of demographic parameters and thus this study provides only a partial view of the ecological relationships in the studied landscape. However, our experiments on predation rates at young afforestations of Mediterranean cropland and adjacent open farmland hint local habitat and landscape features that are indicators of predation risk for bird nests. Predation rates were extraordinarily high, particularly at or nearby large plantations, with high numbers of magpies and low tree development, and located in homogenous landscapes dominated by herbaceous crops and pastures with no remnants of semi-natural woody vegetation. Thus, we conclude that (i) tree plantations based on pines should not be favoured, and even be extirpated, in agricultural landscapes that are highly valuable for ground-nesting bird species and open farmland communities (Traba *et al.*, 2006; Sánchez-Oliver *et al.*, 2013). We also conclude that (ii) abundance of magpies should be diminished to reduce predation risk at tree plantations, preferably by measures related to habitat management that enhance predators of magpies such as some raptor species, as sudden extirpation of generalist predators by humans has been shown to be usually ineffective (Martínez-Abraín & Oro, 2013). However, the outcome of magpie management for reducing predation risk at open habitat is less certain. Additionally, (iii) landscape planning should consider restoring patches of semi-natural woody vegetation and hedgerows (Rey Benayas & Bullock,

2012) because our models predict that they have a positive effect on nest survival. We also recommend assessments of real nest predation risk following afforestation in agricultural landscapes to fully understand and, consequently, reduce its impacts on biodiversity, particularly on ground-nesting birds.

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Supplemental on-line material

Figure S1. Location of the study area in central Spain within the Ciudad Real province and distribution of the tree plantations on former cropland that were used to investigate nest predation at the tree plantations (Experiment 1), on open farmland adjacent to tree plantations (Experiment 2) and at both habitat types.

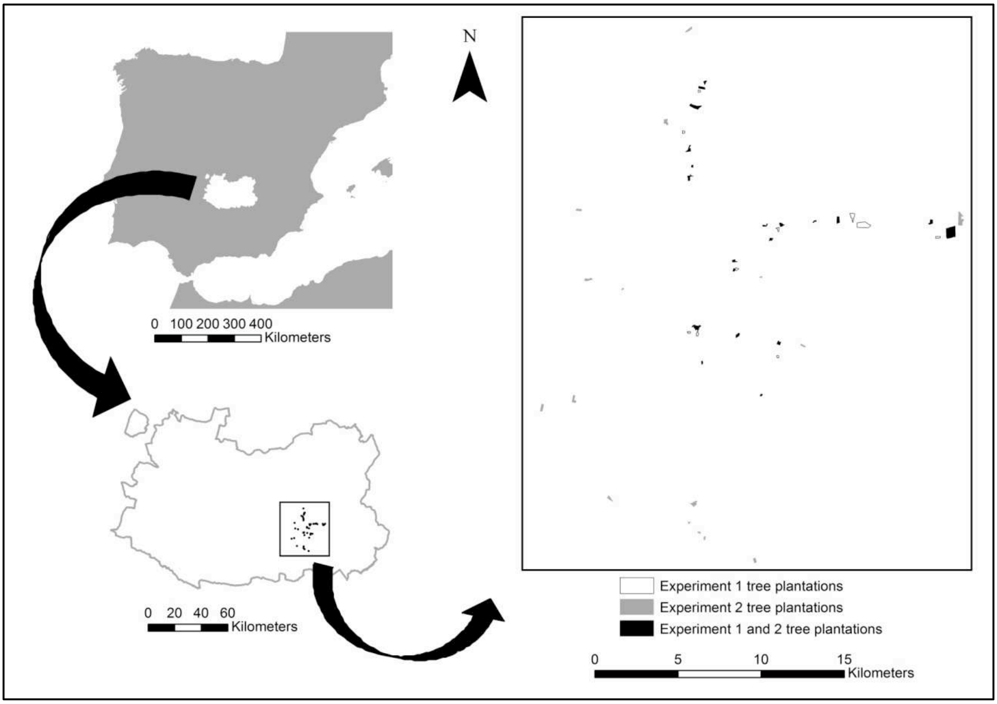

Figure S2. Sketch of the experimental design and associated surveys that were used to investigate nest predation at tree plantations and on adjacent open farmland.

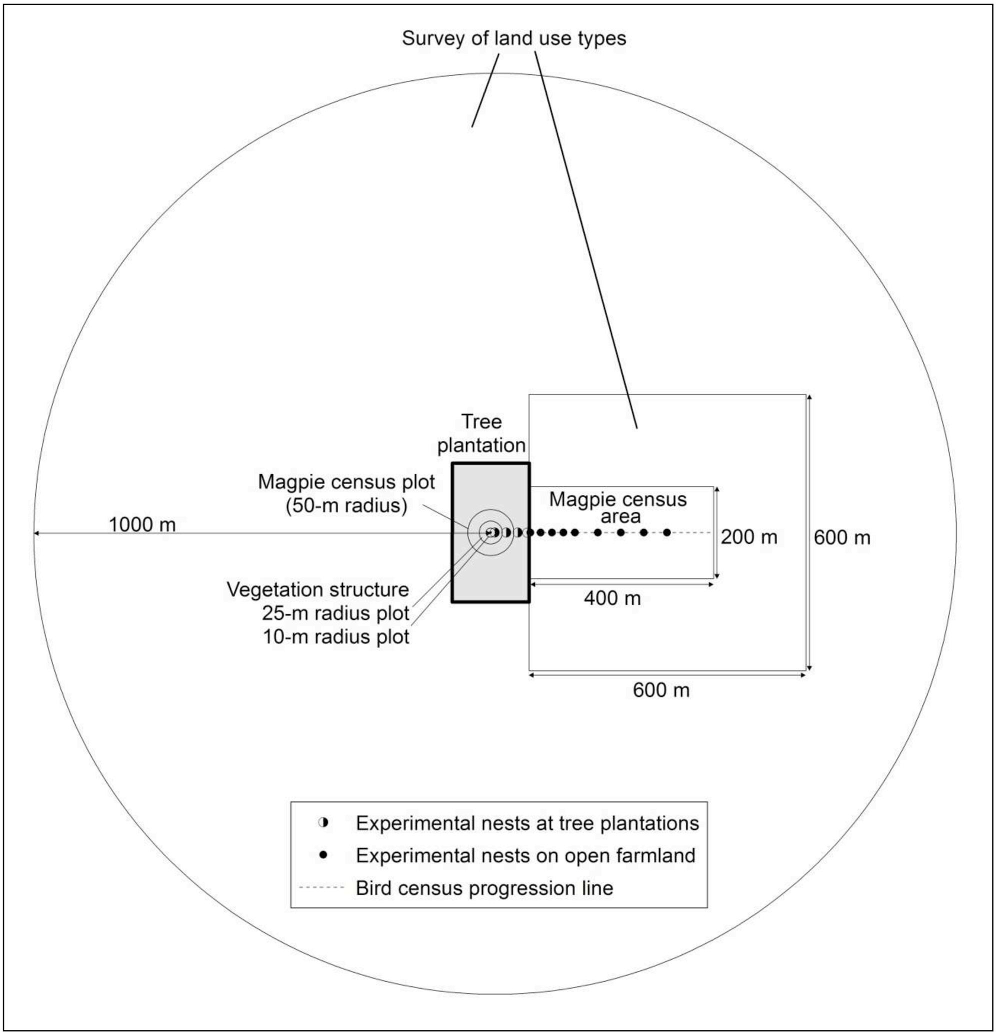

Table S1. Mean, standard deviation (sd) and range (min/max) of the local habitat and landscape variables describing the characteristics of the 30 and 36 studied tree plantations for experiments 1 and 2, respectively. Note: all artificial nests at two out of the 38 tree plantations that were initially selected for Experiment 2 were lost due to ploughing or trampling.

Experiment 1	**Mean**	**SD**	**Range**	
Characteristics of tree plantations				
Area (ha; log-transformed)	1.3	0.8	0.4	3.6
Edge/area ratio	4.7	0.6	4.0	6.2
Position of artificial nests				
Distance to edge (m)	35.4	19.4	25.0	112.5
Height above ground (m)	0.9	0.3	0.4	1.6
Vegetation structure				
Cover of tree layer (%)	36.1	25.5	2.2	100.0
Pine height (m)	3.6	1.6	0.9	7.2
No. Of pine trunks >5 cm dbh	69.7	51.5	0.0	185.0
Cover of shrub layer (%)	6.1	9.8	0.0	46.2
Height of shrub layer (m)	1.3	1.1	0.0	3.3
Cover of herbaceous layer (%)	38.5	37.3	0.0	100.0
Height of herbaceous layer (m)	0.3	0.3	0.0	1.0
Percentage of land use types				
Streams, rivers and lagoons	1.1	1.3	0.0	4.1
Roads and rural tracks	6.7	3.3	0.0	12.0
Urban areas and scattered buildings	2.2	1.9	0.0	7.4
Semi-natural woodland	4.8	5.4	0.3	25.2
Dried-fruit orchards	0.7	3.1	0.0	16.9
Orchards	1.3	1.5	0.0	5.4
Waste lands	7.0	4.1	0.0	14.8
Olive groves	13.6	19.0	0.0	71.2
Pastures with scattered trees	0.4	1.7	0.0	9.4
Scrubland	13.1	7.8	0.0	29.5
Pasture land	1.1	3.4	0.0	19.1
Dry herbaceous cropland	19.0	8.2	0.0	40.4
Vineyard	25.8	12.1	1.0	47.8
Vineyard with olive trees	3.0	3.9	0.0	10.6

Experiment 2	Mean	SD	Range	
Characteristics of tree plantations				
Area (ha; log-transformed)	1.4	0.8	0.3	3.6
Average pine height (m)	3.6	1.4	1.0	6.4
Percentage of land use types				
Streams, rivers and lagoons	0.2	0.5	0.0	2.4
Roads and rural tracks	2.3	2.4	0.0	13.7
Urban areas and scattered buildings	0.4	0.6	0.0	2.3
Semi-natural woodland	0.9	2.2	0.0	9.4
Dried-fruit orchards	1.0	2.3	0.0	9.9
Orchards	0.2	0.8	0.0	4.5
Waste lands	1.3	2.6	0.0	13.0
Olive groves	17.1	14.9	0.0	57.4
Pastures with scattered trees	1.8	5.0	0.0	22.9
Scrubland	0.6	1.7	0.0	11.5
Pasture land	8.7	12.4	0.0	60.0
Dry herbaceous cropland	33.8	27.3	0.6	96.6
Vineyard	29.5	24.4	0.0	82.0
Vineyard with olive trees	0.9	2.7	0.0	10.6
Habitat type where nests were placed	**% of nests**	**No. of nests**		
Olive groves	17.4	45		
Vineyard	17.4	45		
Abandoned cropland and pastures	32.4	84		
Semi-natural woody vegetation	10.0	26		
Dry herbaceous cropland	19.7	51		
Waste lands, roads and rural tracks	3.1	8		

Foto: Alcaraván común (*Burhinus oedicnemus*)

Autor: Juan S. Sánchez Oliver

Capítulo 6

Discusión general

Esta Tesis Doctoral ha investigado los efectos que las plantaciones forestales jóvenes (< 20 años) en tierras agrícolas de ambientes mediterráneos tienen en las comunidades de aves forestales y de aves características de espacios agrícolas abiertos. Estas plantaciones son impulsadas por la PAC y su número previsiblemente pueda aumentar en un futuro próximo. Manejamos la hipótesis de partida de que tanto las características de las plantaciones forestales (por ejemplo, su tamaño y desarrollo) como las de los paisajes donde se ubican (es decir, los diferentes usos del suelo que las rodean) afectan a la comunidad de aves. Esta hipótesis se cumple en general, según indican los resultados obtenidos sobre (1) la composición y la diversidad de la comunidad de aves que coloniza las plantaciones forestales (Sánchez-Oliver, Rey Benayas & Carrascal, 2013; **Capítulo 2**), (2) el test de efectos de tres grupos de factores (efectos aleatorios de la distribución de especies a escala regional, preferencias de hábitat y rasgos autoecológicos) en la ocupación de las plantaciones (**Capítulo 3)**, (3) los efectos sobre la comunidad de aves propia de los medios agrícolas (**Capítulo 4**) y (4) la depredación de nidos de aves tanto en las plantaciones como en los campos agrícolas (**Capítulo 5)**. No obstante, la importancia relativa de las características de las plantaciones forestales respecto a las de los paisajes donde están es diferente en el invierno y en la primavera, enfatizando la idea de que sus efectos no son ni generalizables ni homogéneos estacionalmente. Los resultados obtenidos permiten, además, sugerir algunas recomendaciones con repercusiones sobre la avifauna.

Reforestación de tierras agrícolas y aves forestales de la región

La restauración activa de campos agrícolas mediterráneos por medio de plantaciones forestales incentivadas por la PAC estuvo parcialmente justificada por el objetivo de restaurar antiguos hábitats forestales y su biodiversidad asociada (Comunidad Económica Europea, 1992; Robson, 1997). Dos estudios recientes indicaron que las plantaciones de pinos poseen una mayor riqueza de especies que los matorrales y bosques secundarios que se han regenerado a partir del abandono de tierras agrícolas y pastizales, pero remarcan la incapacidad de estas plantaciones para mantener o incrementar la diversidad de aves especialistas forestales en regiones mediterráneas (Rey Benayas, Galván & Carrascal, 2010; Galván & Rey Benayas, 2011), tal y como sugieren otros autores (Díaz *et al.*, 1998; Maestre & Cortina, 2004; Santos *et al.*, 2006). Sin embargo, todos estos autores estudiaron unos sistemas forestales más maduros y unos paisajes agrícolas más homogéneos que los abordados en este estudio, el cual se refiere a un paisaje agrícola y forestal espacialmente heterogéneo, con abundancia de cultivos leñosos y un archipiélago de plantaciones forestales jóvenes y pequeñas.

Nuestro estudio de la comunidad de aves que colonizan el interior de las plantaciones forestales indica que éstas no constituyen un hábitat adecuado para las aves especialistas forestales de la región (Sánchez-Oliver *et al.*, 2013; **capítulos 2 y 3**). Atribuimos este resultado a varias causas, que incluyen el pequeño tamaño (Sánchez-Oliver *et al.*, 2013), el *pool* regional de especies y su carácter biogeográfico, las características del paisaje alrededor de las plantaciones y el riesgo de depredación. La ocupación de las plantaciones por parte de las aves es explicada por efectos aleatorios de la distribución de especies a escala regional ("random sampling"), más que por los procesos ecológicos relacionados con las preferencias del hábitat o los rasgos autoecológicos de las especies (**Capítulo 3**). La pequeña influencia de estos rasgos en la probabilidad de ocupación de las plantaciones forestales se debe en gran parte a su localización en una región con una avifauna forestal pobre dominada por especies de origen mediterráneo con marcadas preferencias por medios forestales y arbustivos esclerófilos (Monkkonen, 1994; Tellería & Santos, 1994; Suarez-Seoane, Osborne & Alonso, 2002; Carrascal & Díaz, 2003). Así, el conjunto regional de especies forestales limita las posibilidades de colonización de estas plantaciones de coníferas. La mayoría de las especies presentes en la región presentaron abundancias muy bajas o están completamente ausentes en las plantaciones estudiadas, que son colonizadas por especies forestales generalistas, ubiquistas y de amplia distribución, como son el carbonero común *Parus major,* el mosquitero común *Phylloscopus collybita,* el jilguero *Carduelis carduelis,* la paloma torcaz *Columba palumbus,* la urraca *Pica pica* o el estornino negro *Sturnus unicolor,* entre otras, que no contribuyen a aumentar la diversidad regional (Sánchez-Oliver et al., 2013; **capítulos 2 y 3**). Estos resultados coinciden con los de otros trabajos realizados en plantaciones forestales en zonas agrícolas localizadas en la región mediterránea (Díaz *et al.*, 1998; Santos *et al.*, 2006; Razola & Rey Benayas, 2009; Rey Benayas *et al.*, 2010; Silva *et al.*, 2012) y en diferentes regiones de Europa (Wilson *et al.*, 2006, 2012; Smith *et al.*, 2007; Sweeney *et al.*, 2010a, 2010b; Coote *et al.*, 2013).

Las características de las plantaciones forestales y los usos de suelo alrededor de ellas que mostraron una mayor influencia en la composición y densidad de especies local en el interior de las mismas fueron, principalmente, el desarrollo del estrato arbóreo en las plantaciones (frente a los de carácter arbustivo y herbáceo) y los usos del suelo afectados por una profunda antropización de carácter urbano. La mayor importancia de la cobertura urbana alrededor de las plantaciones forestales en la densidad de especies durante el periodo reproductor apunta a la capacidad de atracción de estas pequeñas masas forestales de las especies explotadoras urbanas del centro de España, como son la paloma doméstica o bravía *Columba livia*, la tórtola turca *Streptopelia decaocto*, el verderón común *Carduelis chloris*, el gorrión común *Passer domesticus*, la urraca o el estornino negro. Esta capacidad de atracción incrementa el impacto que el desarrollo urbano tiene en los hábitats circundantes y, por tanto, en la comunidad de aves, especialmente por la influencia de unas pocas especies urbanas muy comunes (por ejemplo, Findlay & Houlahan, 1997; Sauvajot *et al.*, 1998; Odell & Knight, 2001; Palomino & Carrascal, 2007).

El riesgo de depredación es un factor limitante para las especies que pueden habitar estos medios forestales. Así, diversos estudios han mostrado que las plantaciones forestales en medios agrícolas son usadas como refugio por depredadores generalistas (Virgós, Tellería & Santos, 2002; Barea-Azcón *et al.*, 2006; Pita *et al.*, 2009; Fandos, Fernández-López & Tellería, 2012). Uno de estos depredadores es la urraca, que presenta una elevada abundancia en las plantaciones estudiadas (Sánchez-Oliver et al., 2013; **capítulos 2 y 5**) y tiene un efecto significativo en las tasas de depredación de nidos artificiales (**Capítulo 5**). Estos resultados concuerdan con la revisión de Batáry & Báldi (2004), quienes estudiaron las tasas de depredación de nidos en fragmentos forestales y en campos agrícolas, encontrando que el paisaje tiene un efecto relevante sobre estas tasas.

Reforestación de tierras agrícolas y aves características de medios agrícolas

En general, nuestros resultados indican un efecto perjudicial, aunque no muy intenso y diferente en el invierno y durante el periodo reproductor, de las plantaciones forestales estudiadas sobre la comunidad de aves de zonas agrícolas que rodean a las plantaciones (**Capítulo 4**). Este efecto se manifiesta en la repulsión de algunas especies y, sobre todo, en las tasas de depredación de nidos (**Capítulo 5**). Así, la frecuencia de aparición de algunas especies como son la paloma doméstica *Columba livia*, la urraca *Pica pica*, la ganga ortega *Pterocles orientalis* o el verdecillo *Serinus serinus* disminuyó en invierno cerca de las plantaciones. Sin embargo, la frecuencia de aparición de otras especies, como son la perdiz roja *Alectoris rufa,* la tórtola turca *Streptopelia decaocto* o el estornino negro *Sturnus vulgaris*, disminuyó cerca de las plantaciones de mayor tamaño tanto en el invierno como en el periodo reproductor (**Capítulo 4**). Este efecto negativo puede ser el responsable de la ausencia de ciertas especies consideradas comunes en estos ambientes, como son el bisbita campestre *Anthus campestris,* la terrera común *Calandrella brachydactyla,* la cogujada montesina *Galerida theklae,* la totovía *Lullula arbórea,* la collalba gris *Oenanthe oenanthe* y la curruca zarcera *Sylvia communis,* entre otras. La frecuencia de algunas de las especies detectadas respecto a la distancia al borde de la plantación no fue la esperada. Así, los censos del interior de las plantaciones (**Capítulo 2**) y cerca de las mismas (**Capítulo 5**) resultaron en una abundancia de la urraca diferentes a los resultados obtenidos para ella en los transectos en campos agrícolas. Es más, los resultados obtenidos para algunas especies como son el triguero *Emberiza calandra,* la cogujada común *Galerida cristata* o la calandria común *Melanocorypha calandra,* fue contraria, aunque no estadísticamente significativa, a lo encontrado por otros autores (Devictor, Julliard & Jiguet, 2008; Reino *et al.*, 2009, 2013; Morgado *et al.*, 2010). A su vez, la riqueza de especies en invierno disminuyó en las proximidades de las plantaciones y la composición de especies comunes de la comunidad durante el periodo reproductor difirió según la distancia a la plantación más cercana. Así, un resultado relevante de este trabajo es que los predichos efectos perversos de las plantaciones forestales en ambientes agrícolas se diluyen por el pequeño tamaño y desarrollo de éstas y por el carácter de mosaico del paisaje que las incluye.

Se ha demostrado que el incremento de la superficie de hábitats forestales en áreas agrícolas aumenta la abundancia de depredadores de nidos (Andren, 1992; Pita *et al.*, 2009) y, por tanto, las tasas de depredación de nidos de especies de aves que nidifican en el suelo con interés en conservación (Pescador & Peris, 2001; Reino *et al.*, 2010; Suvorov *et al.*, 2012). Los resultados del experimento con nidos artificiales en campos de cultivo adyacentes a plantaciones forestales no sólo corroboran este hecho, sino que reportan unas de las tasas más altas y rápidas de depredación de la literatura científica (alrededor del 90% en dos semanas; **Capítulo 5**). La zona estudiada, caracterizada por ser un medio muy humanizado, con áreas urbanas y presencia de plantaciones forestales, es particularmente favorable para la urraca que, junto a otros depredadores de nidos oportunistas y ubiquistas, implica problemas de conservación para diferentes especies de aves (Andren, 1992; Groom, 1993; Paradis et al., 2000).

Recomendaciones de gestión

Las plantaciones forestales en zonas agrícolas son generalmente controvertidas por el efecto negativo que producen sobre la biodiversidad característica de estos hábitats abiertos, especialmente por su impacto sobre las aves (Butler *et al.*, 2010; Sanderson *et al.*, 2013). Nuestras recomendaciones de gestión son, en síntesis, las siguientes.

(1) En primer lugar, es recomendable que la restauración forestal tenga en cuenta el origen biogeográfico de la avifauna de la región y su capacidad de respuesta para colonizar las plantaciones de coníferas (**Capítulo 3**).

(2) En segundo lugar, las zonas con elevado interés para la conservación de las aves de zonas agrícolas abiertas o esteparias, como las señaladas por Traba et al. (2006) en la Península Ibérica e Islas Baleares, deben ser excluidas de los programas de reforestación de tierras agrícolas e, incluso, algunas ya establecidas deberían ser extirpadas (**capítulos 4 y 5**).

Los diferentes efectos del hábitat local y de las características del paisaje circundante sobre la comunidad de aves encontradas en el interior de las plantaciones hace difícil sugerir prácticas de manejo de las reforestaciones con efectos positivos para todas las especies durante todo el año (Sánchez-Oliver et al., 2013; **Capítulo 2**). No obstante, nuestro resultados apuntan a una serie de medidas de gestión para hacerlas menos perniciosas para la avifauna forestal.

(3) La poda de las ramas bajas y el aclaramiento de estas plantaciones favorecerían el desarrollo del arbolado y producirían un aumento de la complejidad estructural de la vegetación (Sánchez-Oliver et al., 2013; **Capítulo 2**).

(4) La disminución de la abundancia de urracas reduciría los riesgos de depredación en las plantaciones, preferiblemente con medidas relacionadas con el manejo del hábitat que favorezcan a los depredadores de estos córvido, incluidas en una gestión integral del mosaico agrícola y forestal, en lugar de la ineficiente erradicación de las poblaciones de depredadores generalistas (Martínez-Abraín & Oro, 2013).

(5) Adicionalmente, la planificación del paisaje debe considerar restaurar parches de vegetación leñosa semi-natural y setos (Rey Benayas & Bullock, 2012),

ya que los resultados de los experimentos de depredación de nidos predicen que tendrían un efecto positivo en la supervivencia de los mismos (**Capítulo 5**).

Líneas futuras de investigación

A continuación enumeramos unas líneas de investigación que contribuirían a conocer y entender mejor los efectos de las plantaciones forestales jóvenes en campos agrícolas sobre las aves.

(1) El seguimiento a medio y largo plazo de la comunidad de aves de las plantaciones permitiría entender la dinámica de colonización de estos hábitats al aumentar su madurez, aportando mayores conocimientos sobre su relevancia para las especies forestales de la región.

(2) La mayor parte de las investigaciones de esta Tesis han sido de índole fenomenológico. En consecuencia, consideramos interesante realizar experimentos que contemplen la manipulación de la estructura de la vegetación de las plantaciones, su diversificación mediante la plantación o siembra de una variedad de arbustos y árboles y la creación de puntos de agua, entre otros. Ello permitiría testar relaciones causales entre las características de las plantaciones y su avifauna y prácticas de manejo forestal y restauración ecológica para mejorar su biodiversidad y servicios ecosistémicos.

(3) Evaluar el potencial de estas plantaciones para el control de plagas agrícolas basado en aves, ya que las plantaciones podrían ser utilizadas como hábitat de nidificación y refugio para ciertas especies de aves como insectívoras y rapaces que, al menos en parte, se alimentarían en los campos agrícolas circundantes.

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Foto: Sisón común (*Tetrax tetrax*)

Autor: Javi Cáceres

Capítulo 7

Conclusiones

A continuación se enuncian las principales conclusiones de esta Tesis Doctoral. La primera y la séptima son las más generales e inclusivas del resto, por lo que las hemos destacado en negrita y cursiva.

1. Se cumple, en general, la hipótesis de partida de que tanto las características de las plantaciones forestales como las de los paisajes donde se ubican afectan a (1) la composición y la diversidad de la comunidad de aves que coloniza las plantaciones forestales, (2) la avifauna de los medios agrícolas que rodean a las plantaciones y (3) la depredación de nidos de aves tanto en las plantaciones como en los campos agrícolas circundantes.

2. Las plantaciones forestales estudiadas constituyen hábitats de superficie pequeña que atraen a especies forestales generalistas y especies ubiquistas, ampliamente distribuidas y con tendencias poblacionales crecientes en la región; sin embargo, no favorecen a las especies forestales salvo al carbonero *Parus major*.

3. La colonización de las plantaciones de pinos por parte de las especies es explicada sobre todo por la hipótesis del muestreo aleatorio, más que por procesos ecológicos relacionados con las preferencias de hábitat o los rasgos de las especies. La influencia pequeña de estos aspectos autoecológicos se debe en parte a la localización de las plantaciones estudiadas en una región con una avifauna forestal pobre dominada por especies de origen mediterráneo con marcadas preferencias por medios forestales y arbustivos esclerófilos.

4. La comunidad de especies característica de campos agrícolas abiertos está afectada negativamente por estas plantaciones forestales, aunque con efectos distintos en el invierno y durante el periodo reproductor, siendo estos efectos aparentemente atenuados por la heterogeneidad del paisaje. Así, en invierno disminuyó la riqueza de especies y la frecuencia de aparición de varias especies cerca de las plantaciones (por ejemplo, *Pterocles orientalis*), mientras que la composición de especies comunes de la comunidad varió principalmente durante el periodo reproductor. La frecuencia de aparición de otras especies disminuyó cerca de las plantaciones de mayor tamaño tanto en invierno como en el periodo reproductor (por ejemplo, *Alectoris rufa*).

5. Las plantaciones forestales incrementan el riesgo de depredación de nidos, tanto *in situ* como en los campos agrícolas adyacentes, especialmente en paisajes homogéneos dominados por cultivos herbáceos. Las tasas de depredación obtenidas resultaron extraordinariamente elevadas y rápidas, hasta un 88,4% en menos de dos semanas, en comparación con las tasas de depredación de nidos en plantaciones forestales, fragmentos forestales y en zonas agrícolas de diferentes zonas del planeta (valores medios del 59,5%, 66,4% y 60,0%, respectivamente).

6. En su actual estado de desarrollo, estas plantaciones forestales no promueven un incremento de la diversidad regional de aves, por lo que las ayudas de la PAC a la reforestación de tierras agrícolas en ambientes mediterráneos no constituyen una práctica beneficiosa para la biodiversidad.

7. *Planteamos las siguientes recomendaciones relacionadas con la gestión de las plantaciones forestales y del paisaje estudiados: (a) tener en cuenta el origen biogeográfico de la avifauna para la restauración forestal; (b) excluir, e incluso extirpar, la reforestación de tierras agrícolas en zonas con elevado interés para la conservación de las aves características de medios abiertos; (c) podar y aclarar las plantaciones existentes; (d) disminuir la abundancia de urracas, preferiblemente con medidas relacionadas con el manejo del hábitat, que favorezcan a los depredadores de estos córvidos, integradas en una gestión ecosistémica del mosaico agrícola y forestal ; y (e) restaurar parches de vegetación leñosa semi-natural y setos.*

8. Se sugieren tres líneas de investigación futuras para entender mejor los efectos de las plantaciones forestales jóvenes en campos agrícolas sobre las aves: (a) El seguimiento a medio y largo plazo de la comunidad de aves de las plantaciones; (b) la realización de experimentos que contemplen la manipulación de la estructura de la vegetación de las plantaciones, su diversidad y heterogeneidad mediante la plantación o siembra de una variedad de arbustos y árboles, y la creación de puntos de agua, entre otros; y (c) evaluación de su potencial para el control de plagas agrícolas basado en ciertas especies de aves como insectívoras y rapaces que, al menos en parte, se alimentarían en los campos agrícolas circundantes.

Foto de portada: Urraca (*Pica pica*)

Autor: Piet Munsterman

Curriculum vitae

1. Formación académica

Licenciado en Biología.
Facultad de Ciencias de la Universidad de Granada. 2001-2007.

Máster Oficial en Biología de la Conservación.
Facultad de Ciencias Biológicas de la Universidad Complutense de Madrid. 2008-09.

2.- Participación en Proyectos de Investigación y técnicos

Enriquecimiento aviar de plantaciones forestales jóvenes en campos agrícolas meditarráneos.
Fundación Internacional para la Restauración de Ecosistemas. De Marzo de 2012 a Febrero de 2013.

Restauración de la biodiversidad y los servicios ecosistémicos en sistemas agrarios. Un enfoque multi-escala (CGL2010-18312).
Departamento de Ecología de la Universidad de Alcalá. De Febrero de 2011 a Febrero de 2012 y de Abril a Junio de 2013. IP: José M. Rey Benayas.

Efectos de la revegetación (pasiva y activa) en la dinámica y diversidad de plantas leñosas y aves en paisajes agrarios (CGL-2007-60533/BOS).
Departamento de Ecología de la Universidad de Alcalá. De Junio a Noviembre de 2010. IP: José M. Rey Benayas.

Interacciones en los nidos de las aves: Parasitismo de cría y cría cooperativa.
Grupo de Comportamiento y Ecología Animal de la Universidad de Granada. 2010. IP: Manuel Soler Cruz.

3.- Colaboración en Proyectos de Investigación y técnicos

Heredabilidad y repetibilidad de un comportamiento defensivo con base genética. Rechazo de huevos en hospedadores de parásitos de cría.
Proyecto de Excelencia de la Junta de Andalucía. Grupo de Comportamiento y Ecología Animal de la Universidad de Granada. 2007-2010. IP: Juan Gabriel Martínez Suárez.

Conservación de la biodiversidad y gestión del agua en balsas de riego de la agricultura intensiva mediterránea.
Grupo de investigación de Ecología Acuática y Acuicultura de la Universidad de Almería. 2007-2009. IP: Jesús Casas.

4.- <u>Becas disfrutadas</u>

Proyecto de Investigación "Estudio del estado de conservación del fartet (Aphanius iberus) en la cuenca del río Adra (Almería): Diagnóstico actual y previsiones de futuro"
Instituto de Estudios Almerienses de la Diputación de Almería. 2008.

5.- <u>Participación en Seminarios, Congresos, Cursos y en Eventos de Difusión Científica</u>

5.1 Congresos

5.1.1 Presentación oral:

Sánchez-Oliver, J. S. 2013. Influencia de las plantaciones forestales jóvenes en campos agrícolas mediterráneos sobre la comunidad de aves. I Congreso Científico del Programa REMEDINAL-2. Universidad Rey Juan Carlos, Madrid, España, 2013.

Sánchez-Oliver, J. S., Rey Benayas, J. M. y Carrascal, L. M. 2012. Cómo incrementar el valor ornitológico de las plantaciones forestales jóvenes en campos agrícolas mediterráneos. XXI Congreso Español y V Ibérico de Ornitología. Vitoria-Gasteiz, España, 2012.

Casas, J., **Sánchez-Oliver, J. S.**, Sanz, A., Furné, M., Paracuellos, M., Suárez, M. D. & Trenzado, C. 2009. The value of irrigation ponds, as compared with natural habitats, for the conservation of endangered species: the case of the Iberian toothcarp (*Aphanius iberus*, Cuvier & Valenciennes, 1846). Sixth Symposium for Eurepean Freshwater Sciences (SFES6). Sinaia, Romania, 2009.

5.1.2 Póster:

Sánchez-Oliver, J. S., Rey Benayas, J. M. & Carrascal, L. M. 2013. Efecto de las plantaciones forestales jóvenes en campos agrícolas mediterráneos sobre la depredación de nidos de aves. XI Congreso Nacional de la Asociación Española de Ecología Terrestre. Pamplona-Iruña, España, 2013.

5.1.3 Asistencia:

2009. I Congreso Ibérico de Reintroducciones de Especies Silvestres realizado en Jerez de la Frontera (Cádiz).

6.- Publicaciones

6.1 Revistas científicas internacionales incluidas en la base de datos *Science Citation Index*:

Sánchez-Oliver, J.S., Rey Benayas, J.M. & Carrascal, L.M. (2013). Differential effects of local habitat and landscape characteristics on bird communities in Mediterranean afforestations motivated by the EU Common Agrarian Policy. *European Journal of Wildlife Research* In press.

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6.2 Aportación de citas:

Paracuellos, M. & **Sánchez-Oliver, J. S**. 2009. En: Molina, B., Prieta, J. & Lorenzo, J. A. 2009. Noticiario ornitológico. Ardeola 56(1), 2009, 151-172

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6.3 Revistas divulgativas

Sánchez-Oliver, J. S. y Casas, J. 2009. Luces y sombras de la conservación del fartet en el río Adra. Quercus, 284, 34-41.

7.- <u>Formación complementaria</u>

- 2011. Curso “Introducción a la redacción de artículos científicos. I Edición”. Duración: 25 horas. Organizado por la Fundación General UGR-Empresa y el CEAMA.

- 2010. Curso “Metodologías básicas en Ecología Evolutiva y Funcional”. Duración: 40 horas. Organizado por el Departamento de Ecología Funcional y Evolutiva de la Estación Experimental de Zonas Áridas-CSIC.

- 2009. Curso “Análisis de datos ecológicos en R. I Edición”. Duración: 20 horas. Organizado por la Fundación General UGR-Empresa y el CEAMA, Universidad de Granada.

- 2009. Seminario “Introducción a Kepler”. Duración: 3’5 horas. Organizado por el CEAMA, Universidad de Granada

- 2009. Título Propio “Introducción al manejo de los sistemas de información geográfica y su aplicación a la ordenación del territorio”. Duración: 30 horas. Escuela de Posgrado, Universidad de Granada.

- 2009. Taller “Foto-identificación en el estudio de carnívoros amenazados”. Facultad de Ciencias Biológicas, Universidad Complutense de Madrid.

8.- <u>Docencia</u>

- Curso 2010-2011. Colaboración en prácticas en el Grado de Biología. 8 horas.

- Curso 2011-2012. Colaboración en prácticas y seminarios en el Grado de Biología y en el Grado de Ciencias Ambientales. 42 horas.

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